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# Air Pollution and Public Health: A Guidance Document for Risk Managers

Lorraine Craig [a]; Jeffrey R. Brook [b]; Quentin Chiotti [c]; Bart Croes [d]; Stephanie Gower [e]; Anthony Hedley [f]; Daniel Krewski [g]; Alan Krupnick [h]; Michal Krzyzanowski [i]; Michael D. Moran [b]; William Pennell [j]; Jonathan M. Samet [k]; Jurgen Schneider [l]; John Shortreed [a]; Martin Williams [m]
[a] Network for Environmental Risk Assessment and Management, University of Waterloo, Waterloo, Ontario, Canada [b] Air Quality Research Division, Atmospheric Science and Technology Directorate, Environment Canada, Downsview, Ontario, Canada [c] Pollution Probe, Toronto, Ontario, Canada [d] Research Division, California Air Resources Board, Sacramento, California, USA [e] Health Studies and Gerontology, University of Waterloo, Waterloo, Ontario, Canada [f] Department of Community Medicine, University of Hong Kong, Pokfulam, Hong Kong SAR, China [g] R. Samuel McLaughlin Centre for Population Health Risk Assessment, University of Ottawa, Ottawa, Ontario, Canada [h] Quality of the Environment Division, Resources for the Future, Washington, DC, USA [i] World Health Organization, European Centre for Environment and Health, Bonn Office, Bonn, Germany [j] NARSTO, Pasco, Washington, USA [k] Bloomberg School of Public Health, Johns Hopkins University, Baltimore, Maryland, USA [l] Department of Air Quality Control, Austrian FEA, Vienna, Austria [m] DEFRA, London, United Kingdom

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# Air Pollution and Public Health: A Guidance Document for Risk Managers

**Lorraine Craig[1], Jeffrey R. Brook[2], Quentin Chiotti[3], Bart Croes[4], Stephanie Gower[5], Anthony Hedley[6], Daniel Krewski[7], Alan Krupnick[8], Michal Krzyzanowski[9], Michael D. Moran[2], William Pennell[10], Jonathan M. Samet[11], Jurgen Schneider[12], John Shortreed[1], and Martin Williams[13]**

[1]*Network for Environmental Risk Assessment and Management, University of Waterloo, Waterloo, Ontario, Canada,* [2]*Air Quality Research Division, Atmospheric Science and Technology Directorate, Environment Canada, Downsview, Ontario, Canada,* [3]*Pollution Probe, Toronto, Ontario, Canada,* [4]*Research Division, California Air Resources Board, Sacramento, California, USA,* [5]*Health Studies and Gerontology, University of Waterloo, Waterloo, Ontario, Canada,* [6]*Department of Community Medicine, University of Hong Kong, Pokfulam, Hong Kong SAR, China,* [7]*R. Samuel McLaughlin Centre for Population Health Risk Assessment, University of Ottawa, Ottawa, Ontario, Canada,* [8]*Quality of the Environment Division, Resources for the Future, Washington, DC, USA,* [9]*World Health Organization, European Centre for Environment and Health, Bonn Office, Bonn, Germany,* [10]*NARSTO, Pasco, Washington, USA,* [11]*Bloomberg School of Public Health, Johns Hopkins University, Baltimore, Maryland, USA,* [12]*Department of Air Quality Control, Austrian FEA, Vienna, Austria, and* [13]*DEFRA, London, United Kingdom*



**This guidance document is a reference for air quality policymakers and managers providing state-of-the-art, evidence-based information on key determinants of air quality management decisions. The document reflects the findings of five annual meetings of the NERAM (Network for Environmental Risk Assessment and Management) International Colloquium Series on Air Quality Management (2001–2006), as well as the results of supporting international research. The topics covered in the guidance document reflect critical science and policy aspects of air quality risk management including i) health effects, ii) air quality emissions, measurement and modeling, iii) air quality management interventions, and iv) clean air policy challenges and opportunities.**

## INTRODUCTION

### Rationale for the Guidance Document

Air quality projections in several locations in developed and developing countries indicate that pollutant levels may not be significantly reduced over the next 15 to 20 yr. In many cases, sizable expenditures and/or significant societal changes will be required to meet ambient air quality standards.

While there are some uncertainties, there is extensive scientific evidence of population health effects associated with short- and long-term exposure to ambient air pollution, even in areas where the standards are already met. Air quality decision makers are faced with uncertainties concerning the costs of abatement, identifying pollutants and sources that are most harmful, the magnitude of public health benefits associated with emission reduction measures, and the extent to which present-day and future transboundary and intercontinental airflows will compromise local and regional efforts to control air pollution. A more important challenge, however, is that as the more obvious cost-effective emissions control options are implemented, decision makers are faced with uncertainty concerning how to achieve further reductions with the greatest health benefit per unit cost of reduction.

Given the contribution and importance that emissions from local sources have to regional, continental, and global airsheds, it is critical that local emission reduction initiatives are an integral part of national and global clean air strategies. The effectiveness of new market-based mechanisms such as emission trading schemes and legal approaches to air quality management has not been clearly demonstrated. There are opportunities to

Address correspondence to Lorraine Craig, Network for Environmental Risk Assessment and Management (NERAM), University of Waterloo, 200 University Avenue, Waterloo, Ontario N2L 3G1, Canada. E-mail: lvcraig@uwaterloo.ca

achieve sizable co-benefits through joint strategies for greenhouse gas mitigation and air pollutant emission reduction.

Clean air is an important aspect of quality of life. As population growth, urban sprawl, and the number of vehicles and other sources increase, the impacts of air pollution on quality of life become more apparent, including impaired visibility, breathing difficulties among asthmatics and the elderly, and restrictions in outdoor physical activity. Outdoor PM (particulate matter) air pollution is estimated to be responsible for about 4% of adult cardiopulmonary disease (CPD) mortality, about 5% of trachea, bronchus, and lung cancer mortality, and about 1% of mortality in children from acute respiratory infection (ARI) in urban areas worldwide. This amounts to a global estimate of 800,000 (1.2%) premature deaths and 6.4 million (0.5%) lost life years (Cohen et al., 2005). Rising public concern and demand for governments to take further action to improve air quality suggest that guidance to support policymakers in formulating wise air quality management strategies is timely.

This Guidance Document aims to serve as a reference for air quality policymakers and managers and to provide state-of-the-art, evidence-based information on key determinants of air quality management decisions. The document reflects the findings of the five annual meetings of the NERAM (Network for Environmental Risk Assessment and Management) International Colloquium Series on Air Quality Management, as well as the results of supporting international research.

The contributors to the guidance document are recognized experts in the science and policy dimensions of air pollution and health. They represent a range of international perspectives including academia (Daniel Krewski, McLaughlin Centre for Population Health Risk Assessment, University of Ottawa; Jonathan Samet, Johns Hopkins University; Anthony Hedley, University of Hong Kong; John Shortreed, NERAM, University of Waterloo); state and national government organizations (Jeffrey Brook, Environment Canada; Michael Moran, Environment Canada; Martin Williams, UK Environment; Jurgen Schneider, Austrian FEA;, Bart Croes, California Air Resources Board); international organizations (Michal Krzyzanowski, WHO European Centre for Environment and Health; William Pennell, NARSTO); and nongovernmental organizations (Quentin Chiotti, Pollution Probe; Alan Krupnick, Resources for the Future).

## Strategic Policy Directions for Air Quality Management

The NERAM (Network for Environmental Risk Assessment and Management) Colloquium Series on Air Quality Management was launched in 2001 to bring international science, public health, and policy stakeholders together annually to share information and chart a path forward to achieve cleaner air and improve public health. The series was spearheaded by NERAM in collaboration with an international multistakeholder steering committee including representatives from national-level regulatory agencies in Canada, the United States, Europe, and Southeast Asia, as well as international environment and health organizations, industry groups, state and provincial regulators, environmental nongovernmental organizations, and academia. Five annual meetings were held in Canada (University of Ottawa—2001), the United States (Johns Hopkins University—2002), Europe (Rome E Health Authority—2003), Mexico (National Institute for Public Health—2005), and Canada (Vancouver—2006).

The colloquium series over the last 5 yr has seen new and evolving solutions to key issues in air quality risk management and the emergence of a new regulatory paradigm to complement traditional public health standard setting. While air quality standards have historically played and continue to play a central and useful role in regulating air pollutants, the findings of key epidemiological studies suggest that air quality management based on standard setting for single pollutants is simplistic and probably suboptimal in protecting public health. For example, particulate matter mass is a good starting indicator for a broad class of what is recognized to be a serious threat to human health. However, cost-effective air particulate strategies require an understanding of:

1. Local components of the mixture including size and chemical constituents (e.g., ultrafines, organic species, metals).
2. Sources of the various components.
3. Effects on health of the various components, their potential interactions with and synergistic and/or additive effects with gaseous air pollutants, and the benefits likely to accrue from various reductions.
4. The costs of reducing the various components. In certain situations, including so-called "hot spots," the estimated costs of additional abatement requirements to achieve incrementally smaller pollutant reductions to meet air quality standards may outweigh any related public health benefits (Maynard, 2003a; Maynard et al., 2003b; Williams, 2008; Craig et al., 2008).

Underlying these developments is a series of statements that identify strategic directions for air quality management. These statements synthesize the collective thoughts of delegates expressed at NERAM III (Rome 2003), NERAM IV (Mexico 2005), and NERAM V (Vancouver 2006) on future directions for air quality risk management. The statements capture the current thinking of public health organizations (i.e., WHO Regional Office for Europe, UK Environment) and the NERAM Colloquium international planning committee. The statements are summarized in Craig et al. (2007; 2008a; 2008b) and www.irr-neram.ca.

## Structure of the Guidance Document

Innovative approaches that focus on reducing harmful exposures in a cost-effective way are required to make further gains in air quality and public health. The guidance document provides a forward-looking perspective based on lessons

learned and best practice in air quality management to guide decision makers toward the development of cost-effective air quality management strategies.

A conceptual framework for air quality policy development was proposed by NERAM to provide a foundation for the colloquium series presentations and discussions (see Figure 1).

The framework identifies key factors underlying the policy process and illustrates the interplay between scientific assessments of air quality and health effects, policy analysis to assess costs and benefits of proposed options, and aspects of the policy environment (fairness, equity, stakeholder acceptability, technical feasibility, enforceability, government commitment) that influence decision making. The framework recognizes that scientific uncertainty is inherent in the inputs to the decision-making process. The topics covered in the guidance document address the key framework elements.

In the section "Air Quality and Human Health," we review the scientific evidence on the health effects of exposure to ambient air pollution. This section reflects the colloquium series' focus on the health significance of exposures to particulate matter. Evidence from epidemiological, toxicological, and clinical studies in Canada, the United States, Europe, and elsewhere is presented. This section also summarizes new insights from emerging literature and addresses challenges for risk management.

In the section entitled "Emission Inventories, Air Quality Measurements, and Modeling: Guidance on their Use for Quality Risk," we provide an overview of the role of ambient air quality measurement, emission inventories, and modeling in air quality management. This section provides examples from North America and Europe to illustrate the current status, strengths, and limitations of emission inventories, air quality monitoring networks, and air quality modeling activities. Best practice in the development of measurement, monitoring, and modeling capacity for air quality management policy development and policy evaluation is described.

Under "Air Quality Management Approaches and Evidence of Effectiveness," we present strategies for improving ambient air quality at the local, regional, and global levels. Case studies from North America, Europe, and Asia provide examples to illustrate each of the approaches and identify factors associated with successful policy development and implementation. Evidence to demonstrate the effectiveness of various air quality management approaches is presented.

In the section "Emerging Challenges and Opportunities in the Development of Clean Air Policy Strategies," we discuss key emerging issues faced by air quality managers and policymakers with the growing awareness of the health impacts of poor air quality and the increasing costs to achieve further improvements. These issues include the challenges of managing

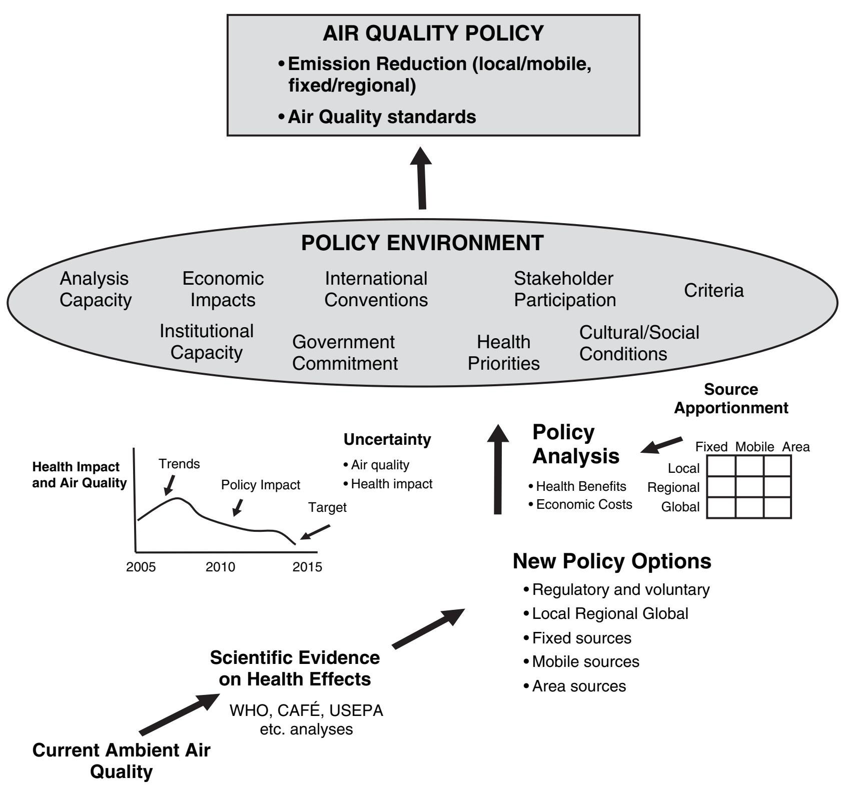



**FIG. 1.** NERAM Air Quality Policy Development Framework.

hot spots and environmental justice and equity considerations. Innovative policy initiatives to complement standards-based air quality management approaches are identified, including integrated strategies oriented toward achieving climate change co-benefits and broader sustainability objectives.

## AIR QUALITY AND HUMAN HEALTH

### Introduction

The primary objective of any air quality management strategy is to protect human health and the environment. From a policymaker's perspective, several key questions on the issue of health effects arise: (1) What is currently known about the impacts of air pollution on public health, (2) which populations are most susceptible, (3) which sources are most damaging to health, and (4) what levels of air pollution are safe and (5) how much health improvement can be expected with air quality improvements? A background paper prepared for the NERAM III Colloquium "Strategies for Clean Air and Health" held in Rome in 2003 framed the discussion of scientific evidence on health effects around these key policy questions. A number of major critical reviews have since been published by the World Health Organization (WHO, 2005a, 2005b, 2005c, 2006a, 2006b), the U.S. Environmental Protection Agency (U.S. EPA, 2004, 2005a, 2006a), and the Air & Waste Management Association (Pope & Dockery, 2006). This section builds on the Rome background paper by presenting new evidence and conclusions from these major reviews.

The focus of this capstone document, as for the NERAM Colloquium series, is on the scientific understanding of outdoor air pollution and its implications for evidence-based risk management. However, there needs to be recognition that air pollution is a broader public health problem with implications for children and adults worldwide. While much of the epidemiological evidence linking air pollution exposures to health impacts focuses on measures of air quality and health in North America and Europe, for millions of people living in developing countries, indoor pollution from the use of biomass fuel occurs at concentrations that are orders of magnitude higher than currently seen in the developed world. Deaths due to acute respiratory infection in children resulting from these exposures are estimated to be over 2 million per year (Brunekreef & Holgate, 2002). While indoor air pollution is responsible for up to 3.7% of the burden of disease in high-mortality developing countries, it is no longer among the top 10 risk factors in industrialized countries in regard to burden of disease. More information about indoor air pollution and its consequences can be found in several recent reviews (WHO, 2002; CARB, 2005).

### Effects of Air Pollution on Population Health

Air pollution is pervasive throughout the world, and represents one of the most widespread environmental threats to the population's health. The World Health Organization (2002) has identified ambient air pollution as a high priority in its Global Burden of Disease initiative, estimating that air pollution is responsible for 1.4% of all deaths and 0.8% of disability-adjusted life years globally. Although the magnitude of the estimated increased risk might appear to be small, the numbers of people affected are large when extrapolated to the entire population.

Despite the seemingly consistent message from the public health community with regard to the need for reduction of risk to the extent possible, there are unresolved scientific issues with attendant uncertainties that are problematic for decision makers. The decision by the U.S. Environmental Protection Agency (U.S. EPA) to retain the annual average standard for $PM_{2.5}$ of 15 μg/m$^3$ averaged over 3 yr, despite the recommendation of U.S. EPA's Clean Air Scientific Advisory Committee (CASAC) for a lower value, is illustrative of how controversy can arise in air quality risk management. In fact with declining air pollution levels in North America and Europe, epidemiological studies become less likely to detect the smaller absolute effects that would be anticipated. As a result, there may be an increased likelihood that any effects that are detected are due to methodologic artifacts as opposed to a true association reflecting upon a possible causal pathway. Uncertainty continues to persist even though many methodological concerns around epidemiological studies have now been addressed and several key reanalyses have been carried out. For example, the extensive reanalysis of two prospective cohort studies, the Harvard Six Cities Study and the American Cancer Society's Cancer Prevention Study II (Krewski et al., 2000, 2004, 2005a, 2005b), confirmed the original findings. Large, pooled time-series studies have also been carried out that produce more precise risk estimates than single-city studies frequently reported in the past (Stieb et al., 2002).

### Scope of Health Concerns

The range of adverse health effects associated with exposure to air pollution has often been depicted as a pyramid (Figure 2). In this formulation, a smaller proportion of the population is affected by the most severe health outcomes such as premature death, hospital admissions, and emergency-room visits; a greater proportion is impacted by conditions that affect quality of life, such as asthma exacerbations that result in work or school absences, and by subclinical effects, such as slowed lung function growth in childhood and accelerated development of atherosclerosis. The range of effects is broad, affecting the respiratory and cardiovascular systems and impacting children, the elderly, and those with preexisting diseases such as chronic obstructive pulmonary disease (COPD) and asthma. The risk for various adverse health outcomes has been shown to increase with exposure and there is little evidence to suggest a threshold below which no adverse health effects would be anticipated (WHO, 2005a).

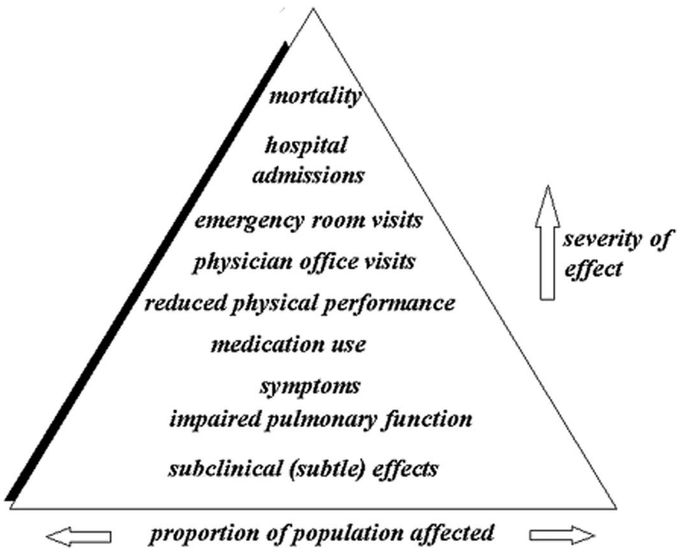



**FIG. 2.** Pyramid of air pollution health effects.

Effects related to short-term exposure
- Lung inflammatory reactions
- Respiratory symptoms
- Adverse effects on the cardiovascular system
- Increase in medication usage
- Increase in hospital admissions
- Increase in mortality

Effects related to long-term exposure
- Increase in lower respiratory symptoms
- Reduction in lung function in children
- Increase in chronic obstructive pulmonary disease
- Reduction in lung function in adults
- Reduction in life expectancy, owing mainly to cardiopulmonary mortality and probably to lung cancer

**FIG. 3.** Health outcomes measured in studies of epidemiological and human clinical studies (WHO, 2006a).

Figure 3 describes the range of health outcomes measured in epidemiological and human clinical studies. The impacts of short-term and long-term air pollution exposures have been studied extensively in North America and Europe for health endpoints toward the peak of the pyramid (i.e., premature death, hospital admissions, and emergency-room visits). More recent studies have examined the health effects of air pollution in low- and middle-income countries where air pollution levels are the highest. The scope of health concerns has broadened from an emphasis on total morbidity and mortality from respiratory causes, such as exacerbations of chronic respiratory diseases, including COPD and asthma, and the respiratory health of children to several adverse cardiac and reproductive outcomes and impacts on susceptible subpopulations, including those with preexisting cardiopulmonary illnesses, children, and older adults. Numerous recent single-city studies have expanded the health endpoints reported to be associated with PM exposures, including (1) indicators of the development of atherosclerosis with long-term PM exposure; (2) indicators of changes in cardiac rhythm, including arrhythmia or ST-segment changes; (3) effects on children and infants; (4) markers of inflammation such as exhaled NO; and (5) effects on organ systems outside the cardiopulmonary systems (U.S. EPA, 2006a). The long-range implications for individuals of some of the intermediate markers of outcome remain to be established, but nonetheless they offer usual indicators of population health.

## Lines of Evidence

Sources of evidence from which to assess the health effects associated with air pollution exposures include observational epidemiologic, toxicological, and clinical studies. The findings of these different lines of investigation are complimentary and each has well-defined strengths and weaknesses. The findings of epidemiological studies have been assigned the greatest weight in standard setting for airborne particles because they characterize the consequences of the exposures that are actually experienced in the community setting.

## Epidemiologic Evidence

The evidence on airborne PM and public health is consistent in showing adverse health effects at exposures experienced in cities throughout the world in both developed and developing countries. The epidemiological evidence shows adverse effects of particles associated with both short-term and long-term exposures. Adverse health effects have been demonstrated at levels just above background concentrations which have been estimated at 3–5 $\mu g/m^3$ in the United States and western Europe for $PM_{2.5}$ (WHO, 2005a).

## Mortality and Long-Term PM Exposure

Associations between air pollution exposure and mortality have been assessed mainly through two types of epidemiological studies. Cohort studies follow large populations for years and typically relate mortality to an indicator of average exposure to PM over the follow-up interval. Time-series studies investigate the association between daily mortality and variation in recent PM concentrations. To establish standards for short-term exposures, regulatory agencies rely on the findings of time-series studies while findings of cohort studies are used to establish annual standards.

Long-term cohort studies of PM and mortality are fewer in number than those of day-to-day variations. They are typically

expensive to carry out and require a substantial number of participants, lengthy follow-up, and information on PM exposure, as well as potential confounding and modifying factors. Most of the studies have been carried out in the United States, but findings have also been reported for two European studies. Two studies of the health effects of long-term exposure to air pollution in large populations have been used extensively in the development of ambient air quality standards for $PM_{10}$ and $PM_{2.5}$.

The Harvard Six Cities Study (Dockery et al., 1993) was the first large, prospective cohort study to demonstrate the adverse health impacts associated with long term air pollution exposures. This study demonstrated that chronic exposure to air pollutants is independently related to cardiovascular mortality. In the group of 8111 adults with 14 to 16 yr of follow-up, the increase in overall mortality for the most polluted city versus the least polluted city was 26%. The range of exposure to PM across the six cities was 11 to 29.6 $\mu g/m^3$ for fine particles.

The American Cancer Society (ACS) established its Cancer Prevention Study (CPS) II in the early 1980s. A subcohort with air pollution data available for counties of residence has been used to assess mortality in relation to air pollution (Pope et al., 1995). The cohort includes approximately 552,138 adults who resided in all 50 states. This study linked chronic exposure to multiple air pollutants to mortality over a 16-yr period. In these two studies robust associations were reported between long-term exposure to $PM_{2.5}$ and mortality (Dockery et al., 1993; Pope et al., 1995).

An independent reanalysis of these two studies was undertaken by the Health Effects Institute in response to industry demands and a Congressional request (Krewski et al., 2000, Pope et al., 2002). The HEI reanalysis largely corroborated the findings of the two studies. In the Six Cities Reanalysis, the increase in all causes of death linked to fine particles was 28% across the pollution gradient from the most to the least polluted city, compared to the original estimate of 26%. For the ACS study, the increased risk of all-cause death associated with fine particles was 18% in the reanalysis, compared to 17% reported by the original investigators. An extended follow-up of the ACS study indicated that the long-term exposures were most strongly associated with mortality from ischemic heart disease, dysrhythmias, heart failure, and cardiac arrest (Pope et al., 2004). For these cardiovascular causes of death, a 10-$\mu g/m^3$ elevation of $PM_{2.5}$ was associated with an 8–18% increase in risk of death. Mortality attributable to respiratory disease had relatively weak associations. Recent analysis of the Los Angeles component of the ACS cohort suggests that the chronic health effects associated with within-city gradients in exposure to $PM_{2.5}$ may be even larger than those reported across metropolitan areas (Jerrett et al., 2005).

An extended analysis to include deaths to the year 2000 confirmed previous findings. The increased risk of all-cause and cardiopulmonary and lung cancer death rose 18 to 30% respectively, though that of lung cancer was 2% (Pope et al., 2002).

Laden's (2006) report on the extended follow-up of the Harvard Six Cities Study found effects of long-term exposure to particulate air pollution that are consistent with previous studies. Total, cardiovascular, and lung cancer mortality were positively associated with ambient $PM_{2.5}$ concentrations. Reduced $PM_{2.5}$ concentrations (mean $PM_{2.5}$ concentrations across the 6 cities were 18 $\mu g/m^3$ in the first period and 14.8 $\mu g/m^3$ in the follow-up period) were associated with a statistically significant reduction in risk for deaths due to cardiovascular and respiratory causes, but not for lung cancer. This is equivalent to a relative risk of 1.27 for reduced mortality risk, suggesting a larger effect than in the cross-sectional analysis. The study strongly suggests that reduction in fine PM pollution yields positive health benefits; however, $PM_{2.5}$ concentrations for the more recent years were estimated from visibility data, which introduces uncertainty into the interpretation of the results of the study.

The Adventist Health and Smog (AHSMOG) study followed cancer incidence and mortality for 6 yr in a group of 6,338 nonsmoking California Seventh-day Adventists, from 1977 to 1987. In 1999, researchers updated the study to follow the group through 1992. In the original analysis, levels of inhalable particles ($PM_{10}$) were estimated. In the update, data from pollution monitors were available. Among men, increased particle exposure was associated with a rise in lung cancer deaths of 138%, and among women exposure was associated with increased mortality from nonmalignant respiratory disease of 12% (Abbey et al., 1999). In 2005, 3239 nonsmoking non-Hispanic White adults who had been followed for 22 yr were examined. Monitoring data were available for both $PM_{10}$ and $PM_{2.5}$. As levels of $PM_{2.5}$ rose, the risk of death from cardiopulmonary disease increased by 42% (Chen et al., 2005).

The relative risk estimates from the major North American cohort mortality studies are summarized in Figure 4 (U.S. EPA, 2006a).

A study involving selected California participants in the first CPS indicated an association between $PM_{2.5}$ and all-cause death in the first time period of the study (1973–1982) but no significant association in the later time period (1983–2002) when $PM_{2.5}$ levels had declined in the most polluted counties. It is noted that the study's use of average $PM_{2.5}$ values for California counties as the exposure indicator likely leads to exposure error as California counties are large and quite topographically variable (Enstrom et al., 2005).

The EPRI–Washington University Veterans' Cohort Mortality Study used a prospective cohort of up to 70,000 middle-aged men (51 ± 12 yr) assembled by the Veterans Administration several decades ago. No consistent effects of PM on mortality were found. However, statistical models included up to 230 terms and the effects of active smoking on mortality in this cohort were clearly smaller than in other studies, calling into question the modeling approach. Also, only data on total mortality were reported, precluding conclusions with respect to cause-specific deaths. A recent analysis of the Veterans' Cohort data reported a larger risk estimate for total mortality related to $PM_{2.5}$ in single pollutant models than reported in the previous analysis. There

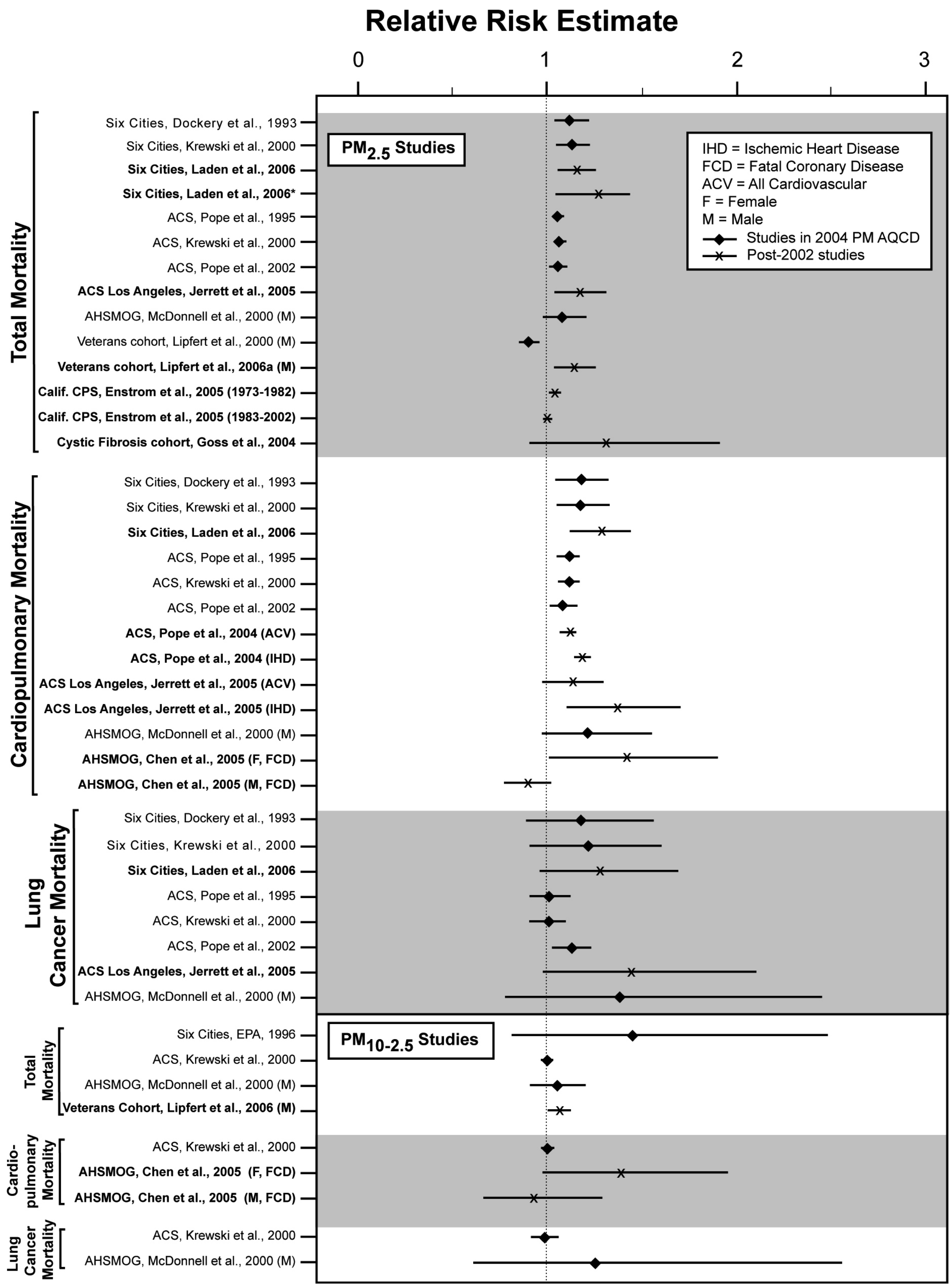



**FIG. 4.** Relative risk estimates (and 95% confidence intervals) for associations between long-term exposure to PM (per 10 $PM_{10-2.5}$) and mortality. *Note the second result presented for Laden et al. (2006) is for the intervention study results (US EPA, 2006a).

was a stronger relationship between mortality and long-term exposure to traffic (traffic density based on traffic flow rate data and road segment length) than with $PM_{2.5}$ mass. In multipollutant models including traffic density, the association with $PM_{2.5}$ was not statistically significant (Lipfert et al., 2006).

A positive but not statistically significant association was reported in a study of a cohort of persons in the United States with cystic fibrosis that focused primarily on evidence of exacerbation of respiratory symptoms. The power of the study to detect an association was limited as only 200 deaths had occurred in the cohort of over 11,000 people. The mean $PM_{2.5}$ concentration was 13.7 $\mu g/m^3$ (Goss et al., 2004).

Further evidence to support an association between long-term air pollution exposure and fatal cardiovascular disease comes from recent cohort studies conducted in Sweden (Rosenlund et al., 2006) and Germany (Gehring et al., 2006). These European studies support U.S. studies and increase confidence in the global applicability of the observations.

## Mortality and Short-Term Exposure Studies

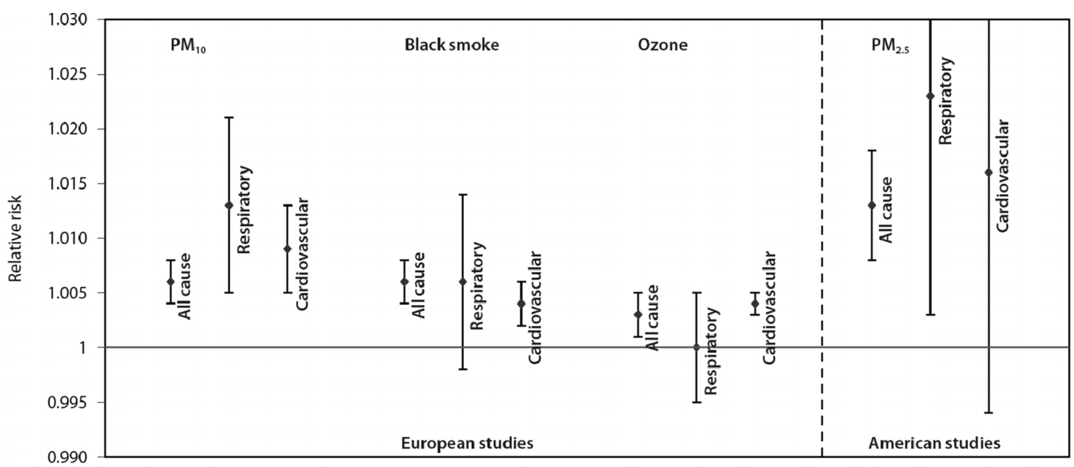
Daily time-series studies examine variations in day-to-day mortality counts in relation to ambient PM concentration measured by air quality monitoring networks. In general, the evidence from daily time-series studies shows that elevated PM exposure of a few days is associated with a small increased risk of mortality. Large multicity studies in Europe (APHEA2 [Air Pollution and Health: A European Approach 2]), and the United States (NMMAPS, based on the largest 90 U.S. cities) indicate that the increase in daily all-cause mortality risk is small but consistent. Concern over the statistical software used in the original analyses prompted a reanalysis of the NMMAPS and APHEA data, along with some other key studies, that was organized by the Health Effects Institute (HEI).

The NMMAPS estimate, based on the largest 90 cities, was revised downward from 0.51% to 0.21% per 10 $\mu g/m^3$ $PM_{10}$ (95% CI, 0.09–0.33) and from 0.51% to 0.31% for cardiorespiratory mortality. The APHEA mortality data reanalysis revealed that European results were more robust to the method of analysis. The WHO meta-analysis estimate (21 of 33 estimates from APHEA2) was 0.6% per 10 $\mu g/m^3$ (95% CI, 0.4–0.8) for daily all-cause mortality and 0.9% for cardiovascular mortality. For $PM_{10}$ and $PM_{2.5}$ the effect estimates are larger for cardiovascular and respiratory causes than for all-cause mortality. The higher European estimates may be due to differences in analytic approaches and other aspects of the methodology, as well as the possibility of a difference in the true effect of PM arising from differing pollution or population characteristics or exposure patterns in the two continents. Figure 5 shows pooled estimates of the relative risks of mortality for a 10-$\mu g/m^3$ increase in various pollutants for all-cause and cause-specific mortality from the meta-analysis of European studies (WHO, 2004).

A review of time-series studies conducted in Asia also indicates that short-term exposure to air pollution is associated with increases in daily mortality and morbidity (HEI, 2004).

## Morbidity

Evidence of associations between exposures and morbidity is complementary to the information on mortality, as it covers a broad range of adverse health effects from changes in biomarkers to clinical disease. Numerous studies have measured the short-term effects of air pollution on morbidity, using clinical indicators such as hospital admissions, counts of emergency-room or clinic visits, symptom status, pulmonary function, and various biomarkers. These studies have included multicity time-series studies (APHEA-2 hospital admission study; NMMAPS) and panel studies of volunteers (PEACE—Pollution Effects on




*Note*: There were not enough European results for a meta-analysis of effects of $PM_{2.5}$. The relative risk for this pollutant is from North American studies and is shown for illustrative purposes only.

**FIG. 5.** Pooled estimates of relative risks of mortality for a 10 $\mu g/m^3$ increase in pollutant from Meta-analysis of European time series studies (WHO, 2006a).

Asthmatic Children in Europe), which have provided data on acute effects on respiratory and cardiovascular systems, and objective measures of lung or cardiac function on a daily or weekly basis, and cross-sectional studies. The case-crossover design has been used to measure risk for acute events, such as myocardial infarction and stroke. In this design, the individual is the unit of analysis and exposures are compared in the "case" period during which the event of interest took place and in one or more "control" periods.

Figure 6 provides a summary of risk estimates for hospital admission and emergency department visits for cardiovascular and respiratory diseases from U.S. and Canadian studies including aggregate results from one multicity study (U.S. EPA, 2006a). There is consistent evidence of increased risk for hospitalization and emergency room admissions for cardiovascular and respiratory diseases. Recent studies, including a new multicity study of 11.5 million people in 204 U.S. counties, provide further evidence of increased risk for cardiovascular and respiratory disease hospitalization related to short-term $PM_{2.5}$ exposure in individuals over 65 yr (Dominici et al., 2006). A number of recent Canadian studies show significant associations between respiratory hospitalization and acute exposure to $PM_{10-2.5}$. For example, studies in Vancouver show increased risk of hospitalization for respiratory illness among children under 3 yr, and for COPD and respiratory in the elderly. Studies in Toronto found an increased risk of hospitalization for asthma in children and associations with respiratory illness in the elderly.

## Public Health Burden of Mortality

Time-series and cohort studies indicate that both short-term and long-term exposures to particulate matter can lead to increased mortality. It is important for public health planning



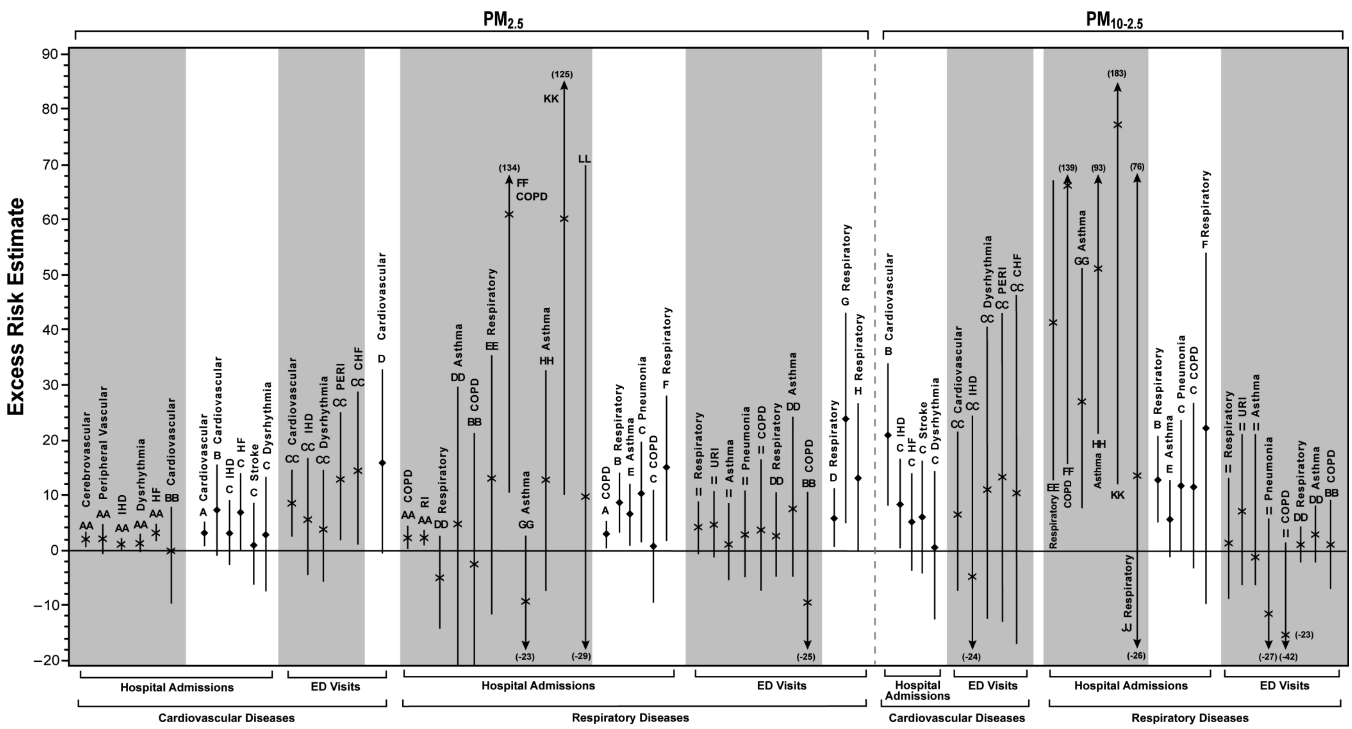



**FIG. 6.** Excess risk estimates for hospital admissions and emergency department visits for cardiovascular and respiratory diseases in single-pollutant models for U.S. and Canadian studies, including aggregate results from a multicity study. PM increment used for standardization was 25 μg/$m^3$ for both $PM_{2.5}$ and $PM_{10-2.5}$. Results presented in the 2004 PM AQCD are marked as ♦, in the figure (studies A through H). Results from recent studies are shaded in grey and marked as × in the figure (studies AA through JJ). (CHF = congestive heart failure; COPD=chronic obstructive pulmonary disease; HF = heart failure; IHD = ischemic heart disease; PERI = peripheral vascular and cerebrovascular disease; RI = respiratory infection; URI = upper respiratory infection) (US EPA, 2006a).

A. Moolgavkar (2003), Los Angeles
B. Burnett et al. (1997), Toronto
C. Ito (2003), Detroit
D. Stieb et al. (2000), St. John
E. Sheppard (2003), Seattle
F. Thurston et al. (1994), Toronto
G. Delfino et al. (1997), Montreal
H. Delfino et al. (1998), Montreal

AA. Dominici et al. (2006), 204 U.S. counties (age >65 yr)
BB. Slaughter et al. (2005), Spokane (age 15+ yr)
CC. Metzger et al. (2004), Atlanta
DD. Slaughter et al. (2005), Atlanta
EE. Chen et al. (2005), Vancouver, Canada (age 65+ yr)
FF. Chen et al. (2004), Vancouver, Canada (age 65+ yr)

GG. Lin et al. (2002), Toronto, Canada (age 6–12 yr, boys)
HH. Lin et al. (2002), Toronto, Canada (age 6–12 yr, girls)
II. Peel et al. (2005), Atlanta
JJ. Yang et al. (2004), Vancouver, Canada (age >3 yr)
KK. Lin et al. (2005), Toronto, Canada (age <16 yr, boys)
LL. Lin et al. (2005), Toronto, Canada (age <16 yr, boys)

to understand the amount of life shortening that is attributable to those premature deaths. Researchers have investigated the possibility that short-term exposures may primarily affect frail individuals with pre-existing heart and lung diseases. Studies by Schwartz (2000), Zanobetti et al. (2000a, 2000b), Fung et al. (2005), reanalysis by Zanobetti and Schwartz (2003), the Zeger et al. analysis (1999), and reanalysis by Dominici et al. (2003a, 2003b) all indicate that that the so-called "harvesting" hypothesis cannot fully explain the excess mortality associated with short-term exposures to particulate air pollution. These studies suggest that any advance of the timing of death by PM is more than just a few days. Brunekreef (1997) estimated a difference in overall life expectancy of 1.11 yr between exposed and clean air cohorts of Dutch men at age 25 using risk estimates from the Dockery et al. (1993) and Pope et al. (1995) cohort studies and life table methods. Similar calculation for U.S. White males yielded a larger estimated reduction of 1.31 yr at age 25 (U.S. EPA, 2004). These calculations are informal estimates that provide some insight into the potential life shortening associated with ambient PM exposures.



## Susceptible Populations

The answer to who is most at risk for PM health effects depends on the level and length of exposure, as well as individual susceptibility. For acute or short-term exposures to moderately elevated PM concentrations, persons with COPD, influenza, and asthma, especially among the elderly or very young, are most likely to be susceptible. Although there may be broad susceptibility to long-term repeated exposure, the cumulative effects are most likely to be observed in older age groups with longer exposures and higher baseline risks of mortality (Pope & Dockery, 2006). Recent work suggests that effects on life expectancy are not uniformly distributed but depend on factors such as educational attainment and socioeconomic status (Krewski et al., 2000), suggesting that life expectancy could be reduced among disadvantaged population groups (Brunekreef & Holgate, 2002).

## Toxicity of PM Components

The question of which air pollutants, sources, or combinations of pollutants are most responsible for health effects is still unresolved. The literature provides little evidence of a single source or well-defined combination of sources most responsible for health effects. With respect to particle size, the epidemiological, physiological, and toxicological evidence suggests that fine particles ($PM_{2.5}$) play the largest role in affecting human health. These particles are generated by combustion processes and can be breathed deeply into the lungs. They are relatively complex mixtures including sulfates, nitrates, acids, metals, and carbon particles with various chemicals adsorbed onto their surfaces. The roles of coarse particles and ultrafine particles are yet to be fully resolved, as are the roles of atmospheric secondary inorganic PM. Other characteristics of PM pollution that are likely related to relative toxicity include solubility, metal content, and surface area and reactivity.

## Role of Gaseous Copollutants

A major methodological issue affecting epidemiology studies of both short-term and long-term exposure effects relates to the use of appropriate methods for evaluating the extent to which gaseous copollutants (e.g. $O_3$, $NO_2$, $SO_2$, CO), air toxics, and/or bioaerosols may confound or modify PM-related effects estimates (U.S. EPA, 2004). Gaseous copollutants are candidates for confounders because all are known to cause at least some adverse health effects that are also associated with particles. In addition, gaseous pollutants may be emitted from common sources and dispersed by common meteorological factors. For example, both CO and particles are emitted from motor vehicles, and $SO_2$ and $PM_{2.5}$ are both emitted from coal-fired power plants. Krewski et al. (2000) found significant associations for both PM and $SO_2$ in their reanalysis for the Health Effects Institute of the Pope et al. (1995) study. Numerous new short-term PM exposure studies not only continue to report significant associations between various PM indices and mortality, but also between gaseous pollutants and mortality. In some cities the estimated PM effect is relatively stable when the copollutant is included in the model, whereas the estimated PM effect in other cities changes substantially when certain copollutants are included. Despite continuing uncertainties, the evidence overall tends to substantiate that PM effects are at least partly due to ambient PM acting alone or in the presence of other covarying gaseous pollutants (U.S. EPA, 2004).

## New Insights

The body of epidemiological, toxicological, and clinical evidence on health effects has strengthened considerably over the past few years. A number of areas of advancement in the understanding of PM health effects have emerged (Chow, 2006). While new studies provide important insights, in general they support previous evidence regarding health effects of air pollution exposures (U.S. EPA, 2006a).

*Cardiovascular effects.* While earlier research focused on the respiratory effects of PM exposure, evidence on cardiovascular outcomes has grown rapidly since 2000. A scientific statement published by the American Heart Association in 2004 indicated concern that the association of airborne particles with adverse cardiovascular outcomes is causal (Brook et al., 2004a). Recent epidemiological, clinical, and toxicologic studies report new evidence linking long-term exposure to fine particles with the development of atherosclerosis. A meta-analysis of cardiovascular hospitalization studies in Europe and the United States consistently shows an increase in relative risk of cardiovascular hospitalizations associated with increments of 10 $\mu g/m^3$ and 20 $\mu g/m^3$ $PM_{2.5}$ (Pope and Dockery, 2006). Numerous new studies have reported associations between ambient $PM_{2.5}$ and subtle cardiovascular effects such as

changes in cardiac rhythm or heart-rate variability (U.S. EPA, 2006a). An extended follow-up of the Harvard Six Cities adult cohort study found that cardiovascular (and lung cancer) mortality was associated with $PM_{2.5}$ exposure (Laden et al., 2006).

*Mechanisms of effect.* Substantial progress has been made in understanding the biological and chemical mechanisms and pathways by which PM causes adverse effects on human health. Recent research has increased confidence that PM–cardiopulmonary health effects observed in epidemiologic studies are "biologically plausible." Figure 7 indicates the various hypothetical pathways of effect that have been explored. Much remains to be learned; however, it appears from human and animal experimental studies that multiple pathways linking exposure to cardiopulmonary health effects are involved, with complex interactions and interdependencies. While the evidence is still evolving and is not yet definitive, there is some evidence to suggest that PM exposure is associated with increased heart rate and reductions in heart-rate variability, suggesting adverse effects on cardiac autonomic function. Other studies have observed, but not consistently, pulmonary or systemic inflammation and related markers of cardiovascular risk such as cardiac arrhythmia, blood pressure changes, arterial vasoconstriction, ST-segment depression, and changes in cardio repolarization (Pope & Dockery, 2006). It is hypothesized that low- to moderate-grade inflammation induced by long-term chronic PM exposure may initiate and accelerate atherosclerosis. Short-term elevated exposures and related inflammation may increase the risk of making atherosclerotic plaques more vulnerable to rupture, clotting, and eventually causing heart attack or stroke.

Exacerbation of existing pulmonary disease, oxidative stress and inflammation, changes in cardiac autonomic functions, vasculature alterations, translocation of PM across internal biological barriers, reduced defense mechanisms, and lung damage have all been related to different levels of PM exposure, as well as to different particle sizes and compositions.

*Local level mortality risk.* An analysis of ACS data that focused on neighborhood-to-neighborhood differences in urban air pollution in Los Angeles using more precise exposure assessment methods found death rates from all causes and cardiopulmonary diseases at least two times higher than previously reported in analyses of the ACS cohort (Jerrett et al., 2005). The highest estimated from original ACS study (Pope et al., 2002) for all-cause mortality was 6%. Taking into account neighbourhood confounders, the risk was about 11%. The annual average level of $PM_{2.5}$ in the most contaminated area was about 24 μg/m$^3$.

*Risks to diabetics.* There is growing evidence to suggest that people with diabetes are more sensitive to cardiovascular effects from air pollution (Jerrett et al., 2005; O'Neill et al., 2005; Zanobetti et al., 2001; Goldberg et al., 2001). Goldberg et al. (2006) reported significant associations between $PM_{2.5}$ and diabetes deaths, as well as total mortality in people with previous diagnoses of diabetes. The acute risk for cardiovascular events for patients with diabetes mellitus may be twofold higher than for nondiabetics. A study of Boston-area residents found that blood vessel reactivity was impaired in people with diabetes on days when concentrations of particles from traffic and coal-burning power plants were elevated (O'Neill et al., 2005). These findings are of particular concern given the

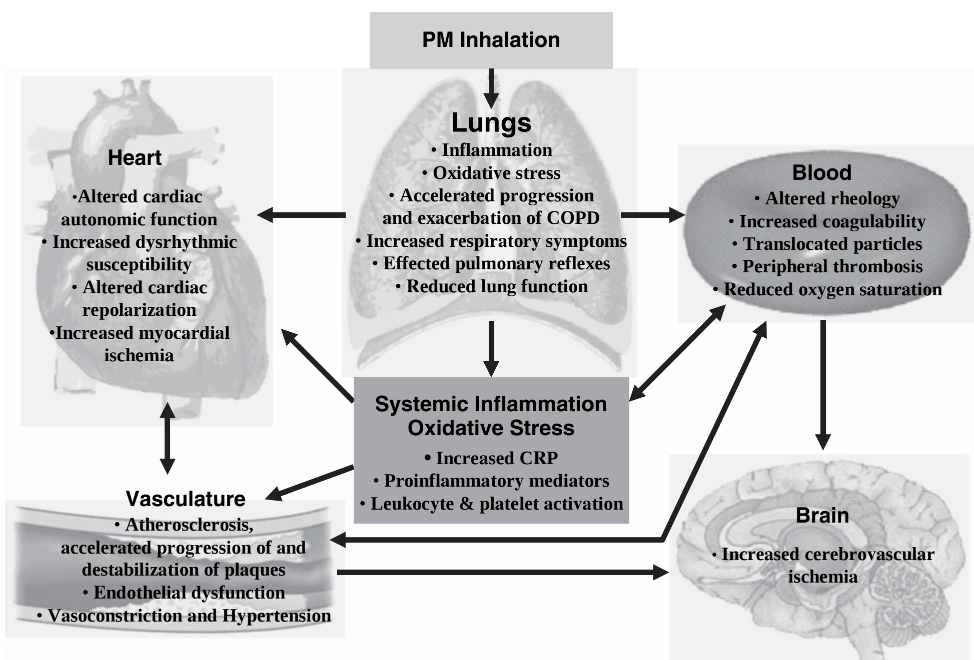



**FIG. 7.** Hypothesized pathophysiological pathways linking PM exposure with cardiopulmonary morbidity and mortality (Pope & Dockery, 2006). Copyright 2008 Air and Waste Management Association. Reproduced with permission of Air and Waste Management Association.

increasing incidence of diabetes in North America. A recent study has indicated mechanistic evidence for diabetes-related susceptibility (Proctor et al., 2006).

*Risks to children.* There is substantial evidence to indicate that PM exposure in children is associated with adverse effects on lung function, aggravation of asthma, and increased incidence of cough and bronchitis. In addition, there is evidence to suggest a casual relationship between particulate air pollution and postneonatal respiratory mortality. Studies on birth weight, preterm births, and intrauterine growth retardation also suggest a link with air pollution, but these studies are not sufficient to draw conclusions about causality (WHO, 2005b).

*Traffic exposures.* Recent evidence has shown that exposures of people living near busy roads are insufficiently characterized by air pollution measurements obtained from urban background locations (Finkelstein et al., 2004; Jerrett et al., 2005; Janssen et al., 2003). In some cities, a significant part of the urban population may be affected by roadway sources. In some urban areas, elevated exposures may particularly affect socially disadvantaged groups (Finkelstein et al., 2003, 2005; Gunier et al., 2003). An analysis of the Veterans' Cohort data reported a stronger relationship between mortality with long-term exposure to traffic than with $PM_{2.5}$ mass (Lipfert et al., 2006).

*Thoracic coarse particles.* While the 2004 U.S. EPA air quality criteria document concluded that there was insufficient evidence of an association between long-term exposure to thoracic particles ($PM_{10\text{-}2.5}$) and mortality, the ASHMOG study (Chen et al., 2005) and Veterans' Cohort study (Lipfert et al., 2006) provide limited suggestive evidence for associations between long-term exposure to $PM_{10\text{-}2.5}$ and mortality in areas with mean concentrations from 16 to 25 $\mu g/m^3$. The extended analyses of the Six Cities and ACS cohort studies did not evaluate linkages between health effects and exposure to $PM_{10\text{-}2.5}$. Recent epidemiologic studies strengthen the evidence for health effects associated with acute exposure to thoracic coarse particles. New toxicology studies have demonstrated inflammation and other health endpoints as a result of exposure to thoracic coarse particles. Clinical exposure studies show changes in heart rate and heart-rate variability measures among exposed healthy and asthmatic adults. It appears that the observed responses may be linked to endotoxins and metals.

## Conclusions

Expanded analyses of ongoing cohort studies continue to provide evidence of associations between long-term exposures to fine particles and mortality (10 $\mu g/m^3$ $PM_{2.5}$ is associated with an approximately 6 to 17% increase in relative risk of mortality, with some outliers). Mixed results have been seen in the AHSMOG study and Veterans' Administration cohort study and the California ACS study.

Understanding the shape of the concentration-response function and the existence of a no-effects threshold level has played a key role in setting air quality standards. Recent empirical evidence concerning the shape of the PM concentration-response function is not consistent with a well-defined no-effect threshold.

Previous cohort studies may have underestimated the magnitude of mortality risks. PM mortality effects estimates tend to be larger when exposure estimates are based on more focused spatial resolution and/or when local sources of exposure, especially traffic sources, are considered.

The available evidence suggests a small increase in lung cancer risk due to combustion-related ambient PM air pollution. The extended follow-up of both the ACS and Harvard Six Cities cohort studies observed PM lung cancer associations, which were statistically significant in the ACS study. Outdoor air pollution typically includes combustion-generated respiratory carcinogens.

Multicity time-series studies in North America and a meta-analysis of European time-series studies support single-city study evidence of an adverse effect of daily $PM_{10}$ exposures on short-term mortality at current concentrations (a 10-$\mu g/m^3$ $PM_{2.5}$ or 20-$\mu g/m^3$ $PM_{10}$ increase is associated with a 0.4% to 1.5% increase in relative risk of mortality).

While earlier studies focused on evidence of respiratory effects, studies emerging over last 10 yr have found a link between both short-term and long-term exposure to particulate matter and risk of cardiovascular disease and death.

With respect to acute or short-term exposures to moderately elevated PM concentrations, persons with chronic cardiopulmonary disease, influenza, and asthma, especially the elderly or very young, are most susceptible. A number of indicators of susceptibility have been identified, including preexisting respiratory or cardiovascular disease, diabetes, socioeconomic status, and educational attainment.

PM exposure impacts the health of children, including deficits in lung function and lung function growth, increased respiratory illness and symptoms, increased school absences, and hospitalizations for respiratory disease. Several recent reviews generally conclude that PM exposure is most strongly and consistently associated with postneonatal respiratory mortality, with less compelling evidence of a link between PM and sudden infant death syndrome (SIDS), fetal growth, premature birth, and related birth outcomes.

Recent research has increased confidence that cardiopulmonary health effects observed in epidemiologic studies are biologically plausible. While a single definitive mechanism has not been, four interrelated pathways involving (a) accelerated progression and exacerbation of COPD, (b) pulmonary/systemic oxidative stress, with inflammation leading to accelerated atherosclerosis, (c) altered cardiac autonomic function, and (d) vasculature alterations have been hypothesized.

There is little evidence of a single major component of PM or a single source or combination of sources that is most responsible for observed health effects; however, epidemiological, physiological, and toxicological evidence suggests that fine particles play a substantial role in affecting human health. The roles of

coarse particles and ultrafine particles are yet to be fully resolved, as are the roles of atmospheric secondary inorganic PM. Other characteristics of PM pollution that are likely related to relative toxicity include solubility, metal content, and surface area and reactivity.

Despite continuing uncertainties, the evidence overall tends to substantiate that PM effects are at least partly due to ambient PM acting alone or in the presence of other covarying gaseous pollutants.

### Issues for Risk Management

The lack of evidence of a threshold concentration for health effects suggests that continued reductions in ambient pollutant levels will result in public health benefits. It further suggests that the target for reduction should be the background concentration. In view of the potentially large costs associated with further abatement measures to achieve cleaner air, questions arise concerning trade-offs between expenditures on air quality management and other measures to achieve public health benefits. In air pollution hot spots and areas where standards have been achieved, air quality risk management becomes more complex. The last section of this document examines these issues and describes innovative air quality risk management approaches intended to complement the traditional regulatory approach focused on the attainment of national ambient air quality standards.

### Key Messages

- A substantial body of epidemiological evidence now exists that establishes a link between exposure to air pollution, especially airborne particulate matter, and increased mortality and morbidity, including a wide range of adverse cardiorespiratory health outcomes. Many time-series studies, conducted throughout the world, relate day-to-day variation in air pollution to health with remarkable consistency. A smaller number of longer term cohort studies find that air pollution increases risk for mortality.
- Health effects are evident at current levels of exposure, and there is little evidence to indicate a threshold concentration below which air pollution has no effect on population health.
- It is estimated that the shortening of life expectancy of the average population associated with long-term exposure to particulate matter is 1–2 yr.
- Recent epidemiological studies show more consistent evidence of lung cancer effects related to chronic exposures than found previously.
- In general, methodologic problems with exposure classification tend to diminish the risks observed in epidemiological studies so that the true risks may be greater than observed.
- Human clinical and animal experimental studies have identified a number of plausible mechanistic pathways of injury, including systemic inflammation, that could lead to the development of atherosclerosis and alter cardiac autonomic function so as to increase susceptibility to heart attack and stroke.
- The question of which physical and chemical characteristics of particulate matter are most important in determining health risks is still unresolved. There is some evidence to suggest that components related to traffic exhaust and transition metal content may be important.
- Despite continuing uncertainties, the evidence overall tends to substantiate that particulate matter effects are at least partly due to ambient particulate matter acting alone or in the presence of other covarying gaseous pollutants.
- Several studies of interventions that sharply reduced air pollution exposures found evidence of benefits to health. New findings from an extended follow-up of the Six Cities study cohort show reduced mortality risk as $PM_{2.5}$ concentrations declined over the course of follow-up. These studies provide evidence of public health benefit from the regulations that have improved air quality.

## EMISSION INVENTORIES, AIR QUALITY MEASUREMENTS, AND MODELING: GUIDANCE ON THEIR USE FOR AIR QUALITY RISK MANAGEMENT

### Introduction

Emission inventories, air quality measurements, and air quality modeling are scientific cornerstones supporting air quality risk management. Developing and applying these tools, along with source apportionment, which are depicted in Figure 8,

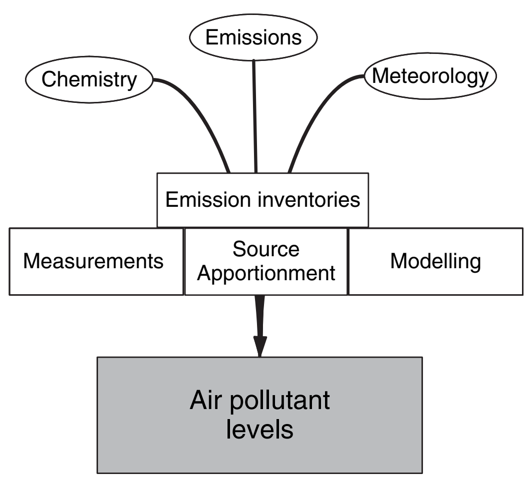



**FIG. 8.** Emission inventories, ambient measurements and air quality models are the tools needed to understand the current air pollutant levels and predict future levels under various policy options.

are the key steps involved in understanding how chemistry, meteorology, and natural and human emissions interact to produce observed levels of outdoor air pollution. In addition, a wide range of air quality (AQ) measurements and exposure analyses is essential for epidemiological research aimed at uncovering the current risks posed by air pollution and for subsequent risk assessment exercises. The purpose of this section is to provide an overview of the roles that emissions inventories, measurements, and models can play in air quality risk management and in understanding air quality issues. This information and the references herein are also intended to provide some insight into current capabilities and best practices associated with developing and applying these essential tools.

Several valuable reports on air pollutant emissions, ambient measurements, and air quality modeling have been published in the past. In particular, the NARSTO particulate matter assessment for policymakers (NARSTO, 2004) describes measurement methods, North American emissions and observations, receptor-based methods of data analysis and interpretation, and the status of air quality models for particulate matter. The World Health Organization report on "Monitoring Ambient Air Quality for Health Impact Assessment" (WHO, 1999) outlines the principles underlying air quality monitoring networks and other related activities (e.g., modeling) that help insure they are of most use for supporting health impact assessment.

Figure 9 shows the basic steps of AQ risk management and specifies how scientific inputs from emissions, measurement, and modeling play a direct role in the policy process. They enable the prediction of air quality improvements associated with emission reduction options, as well as the analysis of the costs and benefits of air quality management options. Although the figure depicts the process in a linear, sequential fashion, with science and policy proceeding separately, in practice the order of steps may be reversed or steps may occur in parallel. In addition, science plays a key role in identifying appropriate air quality goals and options for emission reductions. For example, in developing a conceptual model of the sources and atmospheric processes that lead to current ambient pollutant concentrations, there may be a need to gather additional measurements to test and refine the model before one can thoroughly evaluate whether or not the tools are reliable. In addition, depending upon the maturity of air quality risk management in a particular location, not all steps may be required. Existing measurement programs may be fully adequate, or the AQ models may have already been widely accepted for the intended use.

A crucial step in air quality management is to quantitatively link ambient pollutant concentrations at specific locations or within specific geographic regions to specific emissions (emissions to concentration relationship). This linkage is studied through both receptor and source-based AQ models. Source-based models are capable of predicting future ambient air quality concentrations and are applied to evaluate emission reduction scenarios in the context of Figure 9. Model estimates of concentration changes can then be integrated with concentration-response functions (CRFs) to estimate health benefits.

Table 1 summarizes the various ways in which emissions, measurements and models are applied directly in AQ risk management. Ideally, AQ management should strive to

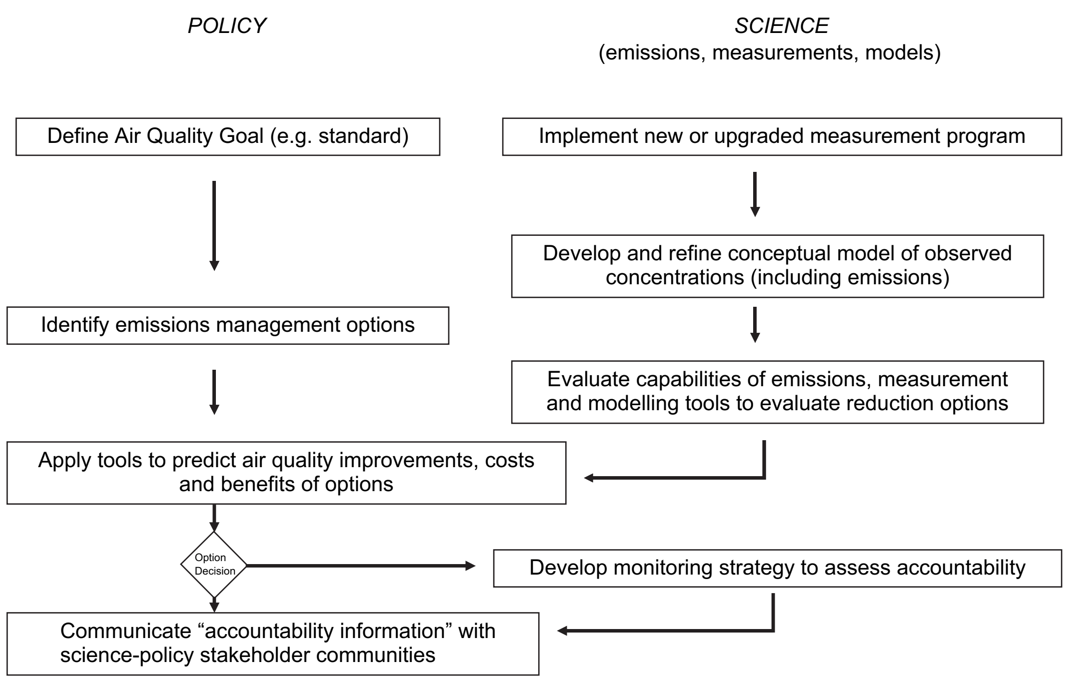



**FIG. 9.** The role of emissions, measurement and modeling in local/regional air quality risk management.

**TABLE 1**
The application of emissions data, air quality measurements and air quality modeling in air quality risk management

| Tool | Area of application in air quality risk management |
|---|---|
| Emissions | • Current emission rates for criteria gases and particles by source type and location serve as the starting point for assessing the need for and feasibility of reductions<br>• Projected emission rates for criteria gases and particles by source type and location and detailed information on the causes of the future changes in emissions<br>• Identification of broad based and detailed emission reduction strategies and technologies by source type and their effectiveness for the emissions of criteria gases and particles |
| Measurements | • Characterization of past and current pollutant levels and identification of exceedances of AQ standards, objectives, or targets<br>• Time series of ambient concentrations at population-based monitoring sites for trend analysis in relation of emission reductions<br>• Determining the relationship between ambient concentrations at population-based monitoring sites and a range of health endpoints (concentration-response function)<br>• The relationship between ambient concentrations of primary and secondary pollutants and emission source categories (source apportionment or receptor models)<br>• Development and evaluation of conceptual models and source-oriented models |
| Models | • Simulation of emission scenarios and quantification of resulting benefits and disbenefits by prediction of ambient concentrations at multiple time and space scales for:<br>  • Base case (e.g., current emissions)<br>  • Emission levels when policies currently "on-the-books" are fully implemented<br>  • New emission reduction scenarios<br>• Estimation of emission changes required to attain AQ objectives or standards<br>• Evaluation of emission estimates<br>• Quantification of source-receptor relationships<br>• Characterization of governing chemical regimes and limiting reactants for current and future conditions<br>• Simulation and design of new or modified measurement systems (network optimization, site selection, input to data assimilation and analysis routines) |



address problems from a multipollutant, risk-based perspective that emphasizes results over process, takes an airshed approach to controlling emissions, creates accountability for these results, and modifies air quality management actions as data on the effectiveness of these actions are obtained (National Research Council, 2004). Although improvements are needed, current emission inventories, measurement activities, and modeling tools are consistent with this objective. To the extent that resources permit, they continually evolve attempting to incorporate the most up to date scientific thinking and technologies, which dictates that to understand and effectively address AQ problems a *one atmosphere* approach is necessary.

Emissions, measurements, and models also play an indirect role in AQ management through the provision of information to the general public or specific stakeholder groups. This includes media reports conveying current pollutant levels or the air quality index (AQI), maps published online (e.g., http://airnow.gov/), and AQ forecasts and/or smog advisories. An example of publicly available emissions information is the North American Commission for Environmental Cooperation (CEC) series of reports ranking major sources and assessing progress (www.cec.org/takingstock/index.cfm).

*Right-to-know* web sites such as the Toxic Release Inventory in the United States (www.epa.gov/tri/) and the Canadian National Pollutant Release Inventory (www.ec.gc.ca/pdb/npri/npri_home_e.cfm) provide specific emissions information for local areas. Public access to emissions information is increasing worldwide (e.g., Mexico: http://app1.semarnat.gob.mx/retc/index.php and www.epa.gov/ttn/chief/net/mexico.html), and international standards for a Pollutant Release and Transfer Register (PRTR) have been established (www.epa.gov/tri/programs/prtrs.htm). This information can be valuable for highlighting areas of large emissions and situations where there has been a lack of progress, as well as for assisting members of the general public in learning about emissions in their region. However, it should be viewed as a starting point for more detailed examination because the information may be outdated, incomplete, and/or misinterpreted by the media or special interest groups.

This section elaborates on the functions of emissions, measurements, and modeling in AQ risk management in

different subsections. The first subsection provides the basics of contemporary emissions inventory development, evaluation, and dissemination. The second subsection describes the various areas of application of measurement data in air quality risk management and provides guidance on technical issues to be considered in establishing a robust measurement program. The third subsection describes the application of models for air quality risk management, identifies key technical issues to consider in using air quality modeling systems, and reviews eight steps for best practice in using models in air quality management. The fourth subsection describes current efforts and future directions to combine the capabilities of emissions and measurement information and air quality modeling to better support air quality management. In the final subsection this section's summary and recommendations are provided.

## Emissions Information for Air Quality Risk Management

### *Introduction*

Without accurate information on the sources of air pollutants—what they are, where they are located, what they are emitting, and how much—it is impossible to identify which sources are most important to control, to predict the air-quality consequences of these emissions, or to monitor the effectiveness of emission reduction programs. Emissions information is provided, assessed, and compiled at many different levels. These can include specific industries continuously measuring and reporting their emissions, national and local governments compiling information and running models to extrapolate the available information across sources, time, and space, and air quality modelers, who process the information for specific applications. To obtain accurate information it is necessary for both government and industry to bear some responsibility, acting in partnership to ensure accuracy, proper interpretation, and continual improvement. Arguably, the emissions information in the United States and perhaps North America (N.A.) represents the current state-of-the-art. Thus, this part of the third section discusses how emissions are determined in N.A. and the current strengths and weaknesses of the available information. Much of the material discussed is from a more extensive report: *Improving Emission Inventories for Effective Air Quality Management Across North America: A NARSTO Assessment* (NARSTO, 2005). This perspective should be relevant to all agencies/stakeholders, regardless of country, seeking to obtain and improve emissions inventories for AQ management purposes.

### *Emission Inventory Development*

Emission inventories are usually developed using the model

$$E = \mathrm{EF} \cdot A \cdot (1 - \mathrm{ER}) \tag{1}$$

where $E$ is the emission rate (e.g., kg/h or tonnes/yr) of a given pollutant, EF and $A$ are the emission factor and activity factor, respectively, and ER ($0 < \mathrm{ER} < 1$) an emission reduction factor, which accounts for any emission control devices that may be applied to the source. The emission factor, EF, is the mass of a given pollutant or chemical species emitted per unit process variable. The activity factor, $A$, is the related process variable such as mass of fuel consumed, vehicle kilometers traveled, etc. in a given amount of time. In reality, emission and emission reduction factors can vary from source to source as well as with the value of the activity factor, type of fuel, operating conditions, age of the source, geographical location, time of year, and so forth. Not all of these complexities can be accurately represented in such a simple relationship, and more sophisticated emission models have been developed for very complex categories such as mobile source emissions (Miller et al., 2006).

Except in rare instances, emission factors or their equivalents are based on measurements. The easiest source class to characterize is large point sources, such as electric generating units or stack emissions from large industrial operations. Emissions from these facilities can be measured by direct sampling of flue gases—as long as reliable sensors and methods are available (they can be in situ or remote) and appropriate sampling techniques are used. For example, when measuring particulate matter emissions it is necessary to mimic the cooling and flue (or exhaust) gas dilution processes that occur immediately after the emissions enter the atmosphere, as many "primary" particles are formed in this near-source region. Using these techniques, gaseous emissions from large point sources, such as $CO_2$, $SO_2$, and $NO_x$, can be estimated to better than ± 20% over time periods as short as 1 h (NARSTO, 2005).

Emissions from more dispersed and numerous individual sources are much more difficult to characterize and evaluate. Examples include fugitive (i.e., inadvertent) emissions from industrial sources, natural emissions from vegetation, agricultural emissions, emissions from small industrial or commercial sources, residential sources (e.g., particulate matter emissions from cooking or space heating), and large-scale biomass burning. For these types of emission sources, direct measurements may be difficult or they may be feasible for only a small sample of the sources in question. All of these factors lead to emission estimates that are more uncertain than for large point sources. These uncertainties can range from a factor of two to complete neglect of an unknown source or chemical precursor that upon later analysis turns out to be significant.

On-road and nonroad mobile sources (automobiles, trucks, aircraft, locomotives, construction equipment, ships, etc.) are a good example of an important, but widely dispersed and variable, component of pollutant emissions. Over the past 50 yr, considerable effort and resources have been expended in several developed countries to develop procedures for estimating emissions from mobile sources. The traditional approach for estimating automobile and truck emissions has been to measure emissions from dynamometer tests of representative vehicles in the laboratory. The dynamometer tests are run to represent typical driving cycles, and the vehicle emissions are measured

in real time. These measurements are used as input to complex mobile source emission models that attempt to simulate vehicle fleet operating conditions for a wide range of urban, suburban, and rural settings. The problems with dynamometer tests are that the number of sampled vehicles may be too small to represent a statistically valid sample, and they may not represent the range of fuels used, driving cycles or conditions, environmental factors, and states of repair of the actual vehicles in use. Consequently, field measurements using instrumented chase vehicles, roadside remote sensing of vehicle plumes, chemical sampling of the air in traffic tunnels, and other experimental setups are used to check and fine tune mobile source emission models. In the future, low-cost portable emission measurement systems (PEMS) and on-board diagnostic sensors (OBDs) may allow cost-effective sampling of a much larger sample of in-service on-road and nonroad vehicles under real-world operating conditions. These data could greatly improve the accuracy of mobile source emission estimates.

Because most sources are not equipped with continuous emission monitors to measure actual emissions, Eq. (1) forms the basis of most data reported in inventories. Although the focus is often placed on the value of the emission or emission reduction factor, the activity factor is equally important. Activity factors can be developed from continuously monitored process data, but as with continuous emission monitors, these data are generally scarce. More frequently, activity factors are developed from economic activity data or activity surveys. Fuel consumption data are a good example of the use of economic activity data that are collected for reasons other than emissions, but can be used in the development of emission inventories. Data are usually available in the United States and Canada by type of fuel used, for various time periods (monthly or sometimes weekly), and for various geographical areas (counties or states/provinces). Information on construction activities can be used to develop emissions from off-road construction sources. Population densities coupled with activity surveys can provide inventory developers with information on emissions from residential woodstove, fireplace, and open burning. Land use maps and satellite data are useful for estimating the types and densities of vegetative cover, which in turn are used to estimate biogenic emissions. In each case, emission estimates can be developed using data that have been collected for other purposes, such as for tax estimation, economic development, or land-use planning.

Projections of future emissions also depend upon these data and estimates in their growth rates in future years. In the absence of significant technological change, past relationships between population growth and the types of activity factors just noted provide a good starting point for estimating future activity levels, and subsequently, future emission levels.

### *Evaluating Uncertainty in Emission Estimates*

Uncertainties are introduced into emission inventories in a number of ways. Emission factors do not usually account for variability in emissions due to changes in source operating conditions, or across the individual sources that make up a source category. New technologies can change processes and emissions, and such changes will not be reflected in emission factors that were based on the original process. Emission factors that are based on idealized operations, such as use of vehicle operating cycles, do not accurately capture actual operation and therefore actual emissions. Measurement biases or errors introduce additional uncertainties into the reported inventory data. These differences can be associated with the location, time, or composition of emissions, leading to uncertainties in the spatial, temporal, or chemical data used in air quality models. Clearly, quantifying uncertainty is an essential “best practice” in inventory development, and it is more efficient to obtain the information needed to assess uncertainty at the time the emissions data are developed as opposed to afterward.

Emission uncertainties tend to be smaller when averaged over larger geographical areas and longer periods of time. Thus, national annual average pollutant concentration estimates are likely to have a lower uncertainty than concentration estimates for a specific urban area over the course of a single day. The differences between actual emissions and estimates based on emission factors are more likely to average out over longer periods of time and broader areas.

As air quality models become more sophisticated to meet the demand of more specific AQ management questions, more detailed emissions information is needed. The ability to model atmospheric processes over the course of a single hour with more detailed chemical reaction mechanisms and in smaller areas means that the differences between the actual emissions within the modeled area and time and the estimate based on an annual average emission factor may be significant. Such discrepancies can result in estimates of pollutant concentrations that do not reflect actual conditions. These differences can lead to misidentification of the most important sources within a given area or erroneous estimates of the specific emissions that need to be controlled for a given source type, as well as incorrect AQ forecasts and population exposure estimates.

Characterizing the uncertainties in non-point-source emission estimates is not a simple task. Two general approaches are usually taken—bottom-up and top-down. In the bottom-up approach, uncertainties (bias and random error) in the individual measurements or parameters that make up the emission model are estimated (e.g., from field measurements similar to those described in the previous section) and represented as a probability distribution function (pdf). These uncertainties are then propagated, often using a Monte Carlo approach, through the model to provide an estimate of the uncertainty in the emission estimate. An alternative bottom-up approach is sensitivity analysis. A simple form of sensitivity analysis is to evaluate the sensitivity of emission-model output to its various input values in terms of the partial derivative of the model output to the input parameter in question. This approach indicates the relative sensitivity of the emission model to its various inputs

and enables a crude estimate of uncertainty by providing a measure of how much the emission estimate would change for a given change in an input parameter.

In top-down evaluations of emission inventories or emission models, ambient measurements or other independent data are used to evaluate the accuracy of the emission estimates. The most effective applications of top-down evaluations are those that are combined with concurrent examination of the original bottom-up inventory data, so that the source of the errors can be identified, rather than simply stating that the inventory is in error (Miller et al., 2006). One top-down method is to compare temporal trends in emission estimates with past trends or to compare trends in the ambient concentrations of a pollutant (or in ratios of pollutants) with the trend in estimated emissions under conditions in which the effects of transport, chemical transformation, and removal can be neglected or accounted for. Results from of this kind of analysis are reported in Parrish et al. (2002) and CRC (2004). A description of this approach is also provided in NARSTO (2005).

Other top-down methods for evaluating emission inventory uncertainties include using alternative methods for estimating emissions (such as comparing vehicular emission estimates based on vehicular distance traveled with those based on total fuel consumption), source apportionment techniques, and inversing modeling. Source apportionment (or receptor modeling) techniques use various multivariate statistical methods to infer source types, source location, and relative contribution from ambient measurements (Watson & Chow, 2005). These methods have been used to evaluate inventories of $PM_{2.5}$ and VOCs (NARSTO, 2004; Watson et al., 2001).

Inverse modeling involves reformulating source-based air quality models so that emission source strengths are expressed in terms of the observed concentrations. In other words, the model is used to deduce the temporal and spatial emissions that explain the observed concentration field. Because of the limited spatial resolution of air quality models, this technique is most commonly used to deduce area sources (Petron et al., 2002; Park et al., 2003; Gilliand et al., 2003).

*Weaknesses of Current State-of-the-Art Emission Inventories*

Over the past 40 yr there has been considerable improvement in the accuracy and completeness of emission inventories, but considerable challenges still remain. As of today, N.A. air quality managers have a good understanding of the emissions from major point sources, and they have used this knowledge in developing effective actions for reducing them. Models for estimating emissions from mobile sources have been continuously improved. The importance of natural and biogenic emissions has been recognized, and this knowledge has affected the design of air quality management strategies in regions where these emissions are significant. In Canada and the United States, emission inventories and models can provide quantitative estimates of emissions at national, state or provincial, and county (or their equivalent) levels for many source categories, and there is an improved understanding of the relative importance of various source categories to specific air quality problems. Air quality managers can use these inventories to track emission trends and to evaluate the effectiveness of measures designed to reduce these emissions. In Mexico, emission inventories have been completed for the Valley of Mexico and the states bordering the United States, and in September 2006 Mexico released its first National Emissions Inventory.

In spite of good progress, emission inventories in N.A. have significant weaknesses that will become increasingly important to address for continued success in dealing with future air quality problems. Recognizing these weaknesses helps provide guidance for improvements and for adopting best practices in inventory development programs in counties with less developed emissions databases. The main weaknesses identified by NARSTO (2005) are:

- Development of mobile source inventories, particularly regarding the speciation of volatile organic compounds, remains a challenging problem. National inventories in Canada and the United States have also indicated problems with the magnitude of carbon monoxide emissions and the temporal trend of nitrogen oxide emissions.
- Emissions for important categories such as biogenic emissions, ammonia, fugitive emissions, open biomass burning, and many other area sources are difficult to determine, and they remain inadequately characterized.
- Emission estimates for air toxics (e.g., the 188 hazardous air pollutants designated by the U.S. EPA) are particularly uncertain since there are so many of these compounds, so many potential sources (many of them area sources), and so little data for establishing emission factors or speciation profiles.
- Emissions of particulate matter and more importantly its chemical constituents, size distribution, and key volatile and semivolatile precursors are in need of improvement across many source categories. Carbonaceous particles (organic and elemental carbon) are a large contributor from many sources and there is limited information for several of them.
- Quality assurance and quality control procedures have not been strictly applied in the development of most North American emission models and inventories. In addition, the documentation of uncertainties and data sources in emission inventories has not been adequate to allow the uncertainties of the entire inventory, or of air quality models using the inventory, to be accurately estimated. These are issues that must be addressed in the initial design of a national inventory development program. Addressing them retroactively is expensive.
- Of necessity, emission estimates must be based on a limited number of emission measurements. If this

number is not representative of real-world activity, the precision and accuracy of estimates developed from these measurements will be limited. More measurements are needed and the issue of representativeness needs to be examined closely.
- The process for developing information on emissions with the kinds of spatial and temporal resolution needed for location-specific air quality modeling and intra-urban scale exposure estimation is problematic and a source of unquantified uncertainty in model results.
- Methods used to estimate emissions of individual chemical species in emission models must be kept up-to-date if they are to provide accurate information.
- Emission inventories must be developed and updated in a timely manner.
- Differences in how emission inventories are developed in adjacent countries create difficulties for jointly managing air quality.



### Actions for Addressing Weaknesses

Reducing known uncertainties in an inventory will provide a more accurate starting point for air quality management strategy development, which should result in more cost-effective approaches. Typically, management actions are initially focused on large point and mobile source emissions. Large point sources are the easiest to characterize and frequently the easiest to control. Mobile sources may be more difficult to completely characterize, but there are few points of manufacture. Thus, control devices can be readily installed during the manufacturing process. As emissions from these sources decline, however, remaining emissions are more evenly distributed across source categories that are even more difficult to characterize, model, and control. These remaining sources will also grow as both population and economic activity increase and errors in emission estimates from smaller individual sources will have greater consequences. These consequences could range from wrongly identifying a pollutant that should be controlled to overlooking source categories whose control could result in more cost-effective emission reductions. As this situation is reached, it becomes increasingly important to address the kinds of emission inventory weaknesses described earlier. Clearly, if these pitfalls are recognized in the initial phases of an inventory development program, it may be possible to avoid them or to address them in a more efficient manner.

The recent NARSTO (2005) assessment provided eight recommendations to the countries of N.A. on how to address the shortcomings of their national emission inventories. The first recommendation was judged to be the most important. The others were ranked as having somewhat lower priority, but in some cases they may also need to be addressed in the course of meeting the first objective. Given the different degrees of inventory development across countries, regions, and even pollutants, the secondary priorities will differ for each situation.

1. *Reduce uncertainties associated with emissions from key under-characterized sources.*
   Comparisons of national emission inventories with ambient measurements and other independent measures should be used to indicate which source categories and pollutants are inadequately characterized and reported. Of particular concern are nonpoint sources including on-road and nonroad mobile sources, as well as fugitive emissions from industrial facilities, landfills, sewage disposal systems, and feedlots. Sources of organic compounds, carbonaceous particulate matter, ammonia, and hazardous air pollutants are typically not well characterized. *Ideally, resources should be targeted to reduce the greatest sources of uncertainty and focused on those source categories (or individual sources and conditions within these categories) whose control will be most effective in reducing costs and health risks while achieving air quality management goals.*
2. *Improve speciation estimates.*
   Contemporary air quality issues, such as particulate matter and ozone and the identification of hazardous air pollutant "hot spots," require detailed information about the species being emitted from the sources. Contemporary emission inventories are weak in this regard. *It is essential that source speciation profiles be continually updated and assessed. In addition, the related activity data must be developed to estimate more accurately speciated emissions of particulate matter and precursors, volatile organic compounds, and toxic air pollutants.*

   Speciation is most important to the management of ozone and fine PM, but it can also be important to air toxics and to some extent climate issues (black carbon vs. organic carbon; different GHGs). More accurate quantification of the species being emitted will result in better air quality modeling results, and in better identification of which sources should be controlled in order to gain the greatest improvements in air quality and in human health, particularly if the more toxic constituents and their sources can be identified.
3. *Improve existing and develop new emission inventory tools.*
   Technical advances in instrumentation and computation are enabling emission measurements and analyses that have been previously impractical. Examples of these innovations include portable emission measurement systems for direct measurement of emissions under real-world conditions and the application of various remote-sensing techniques for measuring emissions or verifying emission estimates. Continuing development of these and other technologies, as well as consideration of different approaches to deriving emission information, is likely to improve emission inventory measurements and analyses. *Funding agencies need to continue to support the development and application of new technologies for measurement of emissions and ambient concentrations of pollutants*. Application of these technologies will assist the development of emission models that more accurately represent emissions from real sources in time and space.

4. *Quantify and report uncertainty.*
Uncertainties in emission estimates and the parameters or data used in emission models must be quantified. Uncertainties in emission inventories, processors, and models of Canada, the United States, and Mexico are poorly documented. As a result, the reliability of emission estimates cannot be quantified. *Quantitative measures of uncertainty and variability must be a standard part of reported emission inventory data. Agencies responsible for producing emission inventories must develop specific guidance on how to prepare and report information on emission uncertainties.*

This recommendation is probably the most difficult to describe and implement. Quantifying the uncertainty of an inventory estimate, and subsequently the estimates of pollutant concentrations, will enable decision makers to assess the likelihood that the desired outcomes will be achieved. A highly uncertain inventory means that the desired air quality improvements are less likely to occur because there is less confidence that inventory estimates accurately describe the actual situation. The ability to assess the potential for strategies to achieve the desired results will help decision makers to determine what steps are most likely to yield improvements.

5. *Increase transnational inventory compatibility and comparability*.
As air quality problems become increasingly global in nature, it is vital that emission inventory development and reporting programs be coordinated internationally. Although there have been substantial improvements in reporting national emission inventories in mutually consistent ways, further work is needed to make these diverse inventories more comparable across organizations, purposes, political boundaries, and time periods. *International standards for emission inventory structure, data documentation, and data reporting should be developed. Such standards are needed to facilitate management of long-range transport and transboundary air quality issues.*

Inventories that are compatible across boundaries, and with inventories of other pollutants (GHGs, toxics, etc.), allow air quality managers to account for emissions that occur outside their domain of responsibility. This reflects the physical reality that pollutants do not recognize political boundaries. Comparability with past inventories enables air quality managers to more accurately assess the degree to which previous air quality management strategies have been successful.

6. *Improve user accessibility.*
The accessibility of emission inventories and emission models is impeded by the sheer size of the files and the cumbersome manner in which the data are reported and archived. *As improved accessibility to emission data is critical to meeting the diverse needs of the user community, increased efforts should be made to facilitate user accessibility to emission inventory data and models through the Internet and other electronic formats.* It is also suggested that emission inventories be made more transparent and easy to update. National inventories do not always contain the most recent emission information. Methods for allowing continuous updating with appropriately validated data from a variety of sources (researchers, industry, government agencies, etc.) need to be developed.

As more groups are able to access inventory data quickly and easily, a better understanding of air quality problems can be expected and more alternative approaches to air quality management can be developed. The more alternative approaches that are available, the more likely it is that air quality will be improved in the most cost-effective manner.

7. *Improve timeliness.*
Timely and historically consistent emission inventories are essential for assessing the current emission environment (and for keeping abreast of economic conditions and changes in technology) and for tracking progress in improving air quality. *Plans and processes need to be put in place for preparing and reporting national emission inventory data on a yearly basis.*

Timely inventory data more accurately reflect the actual situation. Ideally, inventories would be updated almost continuously so that air quality managers could see how the mix of emissions is changing due to external changes (population growth, changes in technologies, economic forces, etc.) and to changes in air quality management strategies. Continuously updated inventories remain far off, but a realistic goal is to minimize the time between when the inventory data are submitted and when they are reported.

8. *Assess and improve emission projections.*
Realistic projections of future emissions are important for developing and assessing strategies for attaining air quality standards and for evaluating future-year effects of new regulations. *Emission projection methodologies for all emission inventory sectors should be evaluated to determine the accuracy of past projections and to identify areas for improvement. Attention should be paid to assuring the compatibility of short-range projections that are more typical of air-quality-related emissions with the long-range projections that are made for climate-change applications. Projections of future emissions are also dependent upon the quality of the base-year emissions. Therefore, realistic projections cannot be made unless these base-year emissions are as accurate as possible.*

Because most air quality management approaches estimate the future effects of air quality management actions, it is important to understand what changes are due to the air quality management strategy and what changes are due to changes in population, technology, etc. Improved projections also provide guidance regarding the level of emission reduction that may be needed to achieve and maintain the desired air quality over the long term.

*Further Issues Regarding Emission Inventory Improvement*

*Scale.* There is a growing demand for reliable small-scale emissions data. They are needed to model urban-scale air quality for the purpose of improving population exposure estimates used in health research and in risk assessment and for control scenarios focused on local emissions (e.g., mobile sources, urban planning). At the local level this issue of scale is significant and a detailed understanding of source variability and how sources differ from the national average need to be included. This involves not only the differences in technologies (processes, control technologies, etc.), but also how the sources operate. If there are significant differences between the local practices and national practices, the inventories will need to account for them. In addition, small sources that are not important at regional and national scales and hence may not be required to report their emissions can be important at the local scale.

Area and mobile sources are generally more important at the local scale, and such changes in the mix of source types can result in air pollution issues at the local scale being different from those at a national or regional scale. A good example is $PM_{2.5}$: In the eastern United States sulfates are the key to $PM_{2.5}$ reductions at the regional scale. At the local scale, however, carbonaceous PM tends to be the issue of most concern, because it is locally emitted (traffic, industries, open burning, and biogenic sources) and because the time scale of $SO_2$ chemistry results in sulfur dioxide being a gas-phase problem locally, but a PM problem farther downwind.

Clearly, developing a local-scale inventory is very demanding. Often such efforts are conducted by local groups based upon their own techniques and/or assumptions. There is a need to consider how such parallel efforts that may involve differing approaches could be standardized and improved, including the assessment of hot-spot emissions, interfacing with exposure models, such as the Regional Human Exposure (REHEX) Model, and intake fraction parameters. This latter parameter is the fraction of the emissions of a pollutant taken in by people (Marshall et al., 2003) and provides a means of weighting sources according to exposure potential. Primary emissions in a rural area and/or from high stacks have a smaller intake fraction than those emitted at ground level in cities, and arguably, for health protection the latter types of sources should be more important to control.

*Groundtruthing efforts.* In current practice, inventories are most effectively used as the starting point for understanding the contributors to air quality problems. They form the basis for air quality modeling and for identifying the sources that are most significant within an airshed. Additionally, best practice dictates that receptor models, fuel-based (vs. vehicle-travel-based) inventories, inverse modeling, and other approaches be used, independently, for inventory verification. This is important to undertake to identify problem areas in the inventories in order to minimize the impacts of erroneous inventory data. These intercomparisons or evaluations may also help identify when the other, perhaps newer, approaches are outperforming the traditional approaches, thus leading to new models, different forms of input data, and better emissions inventories.

*Costs of emission inventory development and improvement* The U.S. federal government currently invests approximately $25 million per year to develop and update emission inventories (NARSTO, 2005). This does not include the amounts spent by state and local agencies (estimated at about $10 million per year) or the additional costs that would be required to address the shortcomings identified in the current inventories, which are estimated to be on the order of an additional $35 million per year (NARSTO, 2005). In Canada, about $6 million (U.S.) per year is invested in compiling its national inventory, not counting local and provincial efforts, and Mexico has spent about $600,000 (U.S.) per year in developing its National Emission Inventory. The costs of addressing emission inventory shortcomings in Canada and Mexico are proportionally similar to those of the United States—about $6 million and $1 million per year, respectively (NARSTO, 2005).

The cost of developing emission inventories is a function of their purpose. The relatively low cost of developing Mexico's national emission inventory is a consequence of its relatively low level of detail and the use of previously existing information. At the other end of the scale, the Electric Power Research Institute (EPRI) spent $50 million to quantify emissions of hazardous air pollutants from electric generation units (EPRI, 1994). The American Petroleum Institute, the U.S. Department of Energy, and others spent about $6 million to measure combustion emissions from refineries. These higher cost examples reflect the greater expense of obtaining highly detailed information, such as speciation of organic and metal compounds, which are present in flue gasses at very low concentrations (NARSTO, 2005).

## Measurement of Ambient Pollutant Concentrations

Along with emissions data, ambient air pollutant measurements are part of the foundation of effective air quality risk management. While all measurements have the potential to play a role, the degree of support varies depending upon a number of factors. The type of measurements and the design of the measurement program are critical factors, but just as important is the amount of time and effort dedicated to analysis and interpretation of the data. When ambient pollutant observations and related measurements are examined to the full extent possible they can support multiple objectives, which includes:

- Describing current risks and detecting potential future risks to human and environmental health.
- Documenting trends in order to demonstrate efficacy of past and present policies (e.g., emission reductions).

- Developing models capable of predicting ambient pollutant concentrations from knowledge of emissions and emission changes.
- Providing information needed to derive quantitative relationships between ambient concentrations and human health (or other adverse effects on the environment, climate, or visibility).

Measurement data and most often routine monitoring data are central to AQ–health studies and the concentration-response functions (CRF) derived from this research. Ultimately, measurements can lead to new insight into the specific pollutants or sources posing the greatest risk to health, which can help in the development of more efficient risk management strategies. This section summarizes a range of issues related to how measurements best support air quality risk management. Protecting human health is one of the main motivations and thus, much of the discussion is geared toward that aspect of air quality measurements. However, other issues related to the measurement objectives just listed are also discussed, including general guidance on technical issues to be considered in establishing a robust measurement program.

*Application to Health Studies*

Exposure to air pollutants has the potential to lead to a variety of adverse health impacts (see section Air Quality and Human Health). Not all such impacts are likely to have been identified and/or adequately characterized, given the diversity of exposure scenarios, the diversity of the population, and the myriad of possible biological pathways. A range of health research approaches and detailed air quality and exposure measurements are thus needed to continue to advance knowledge.

The World Health Organization published a European Series Report on "Monitoring Ambient Air Quality for Health Impact Assessment" (WHO, 1999). The goal of that report was to describe strategies and methods for providing information on ambient air quality that would be adequate for health impact assessment. Many issues that are relevant to this section are discussed, and therefore, the report is recommended background reading.

*Concentration-Response Functions*

For air quality risk management, CRFs based upon direct links between ambient observations of a range of air pollutants, as measured by standard monitoring networks, and acute and chronic impacts occurring within the general population are crucial. Demonstrating such "real world" associations also establishes that air pollutant effects are relevant to actual conditions. This necessitates that ambient measurements continue to be obtained to support both acute and chronic exposure health studies. The level of detail that these measurements should provide, including direct personal exposure studies or other extrapolations related to exposure (e.g., intake fraction, population-weighted concentrations), will depend upon the type of study they are intended to support. There is also considerable demand for better information on the specific pollutants, mixtures, and/or sources that have the greatest impact upon health. This can only be satisfied by enhancing ambient measurement and exposure research activities and fully capitalizing on the recent advances in measurement capabilities (Wexler & Johnston, 2006). In addition, advances in statistical methods that can simultaneously exploit geographic and temporal variations in the interrelationships among pollutants coupled with new monitoring strategies should provide new insights regarding the most harmful pollutants or mixtures.

One of the key challenges in working with ambient pollutant data is to derive CRFs for individual pollutants and for pollutant mixtures that are appropriate for risk management or cost-benefit analysis. They must be scientifically defensible and their uncertainties and/or their strengths and weaknesses need to be understood in detail. Significant challenges arise from confounding and differential exposure error among pollutants. Among other things, an association between a health endpoint and a given pollutant may be because that pollutant is acting as an indicator for an unmeasured pollutant (gas or particle) or mix of pollutants (Brook et al., 2007a). In this situation the relevance of a CRF derived from these data must be questioned and applications (e.g., cost-benefit analysis) must be done with clearly stated caveats. Research is needed to determine the true implications of such caveats and the appropriate interpretation of predicted health benefits.

Conceptually, CRFs should be the same among different urban populations given similar distributions of time activity and susceptible individuals, and, within the range of uncertainties, this is generally the case for PM. However, there remains some variability and the causes have not been resolved. Differences in the air quality data used to derive the CRFs likely play a role in that there are typically variations between how the monitoring sites relate to the population (i.e., their location[s]) and to their actual exposures (e.g., prevalence of air conditioning, which influences indoor penetration, differs geographically). It is also likely that the nature of the confounding among pollutants (measured and unmeasured) and their exposure errors differs among locations or cities. These issues lead to uncertainty in regard to which CRFs should be used to guide risk management. Heterogeneity among risk estimates and possible causes were discussed during the NERAM V Colloquium (Samet, personal communication).

For cost-benefit analysis it is not possible or reasonable to have separate CRFs for every city or population of concern. Meta-analyses have therefore been undertaken to derive "generic" CRFs. This approach assumes, however, that the air pollutant for which the meta-analysis is undertaken is truly the cause of the adverse effects, as opposed to being an indicator for some unmeasured component in the pollutant mixture. If the latter is the case then it is feasible that the relationship between the "indicator" and the true causative pollutant(s) could change

among study locations and hence the "mean" effect from the meta-analysis would not be appropriate. This issue could be relevant with respect to nitrogen dioxide ($NO_2$), for which an association with mortality has been detected in a number of cities (Stieb et al., 2003, Burnett et al., 2004; Samoli et al., 2006).

*Ambient Measurements to Indicate Exposure*

No matter how detailed, ambient measurements cannot reflect what a person or most members of a population are truly exposed to. These measurements are an indicator for some aspect of the air pollutant stresses that the population is confronted with. In reality, the relationship between these indicators and the actual exposures of the population may differ from pollutant to pollutant and from one monitoring site to the next. Site location relative to the population and selection of the pollutants to measure at each site are therefore critical issues.

Acute exposure-effect studies and chronic exposure-effect studies have different requirements regarding site location. The former requires that the measurements accurately reflect temporal variations in the population's exposure, while the latter is interested in how exposure levels vary across space. This could range from differences from one location in a large city to another (intra-urban) or differences between cities (inter-urban). For both types of studies the ideal measurement data are rarely if ever available for both financial and technical reasons. Therefore, compromises are necessary and it is important to understand the limitations of and implication in using the available measurements for health studies or risk assessment.

The best practice for both acute and chronic studies is to measure multiple pollutants at any site that is established and to operate more than one site in the region containing the population of interest. European criteria for site coverage and representativeness are discussed in Kuhlbusch et al. (2004). In their examination of the networks reporting to AirBase they presented an example for the Ruhr district in Germany and assessed differences among countries examining the extent to which monitoring networks were being operated in accordance to the air quality directives with respect to protection of human health. In terms of site locations some of the key criteria to consider in network evaluation are:

- Sites are established that provide data on the highest concentrations.
- The network comprises both hot-spot and urban background sites.
- Hot spot sites are representative for at least 200 $m^2$.
- Urban background sites are representative for several square kilometers.
- Urban background sites are representative for similar locations not in their vicinity.
- Sites are established that are representative for the exposure of the general population.

The spatial distribution of the population, local physical features (e.g., topography, shorelines), and prevailing meteorological conditions are all important to study when selecting site locations or choosing sites to be used in health studies. With this information a better understanding of the link between each set of measurements and the population will be realized and population-weighted concentrations can be derived with more confidence. This perspective also helps identify and address weaknesses in the monitoring network, via new sites or special studies (e.g., saturation monitoring, deployment of mobile labs).

Gaseous air pollutant time series developed for epidemiological research (acute effects) on the Toronto population have been based upon the average hourly concentrations from thre to four different representative sites (e.g., Burnett et al., 1997). Their locations reflect some of the main outdoor environments people commonly encounter, such as residential and commercial areas and near roadway conditions. In many European cities it has been common practice to measure in the urban background and at "curbside" or traffic locations to obtain information on the range of concentrations experienced (Kuhlbusch et al., 2004). Combining data from multiple monitoring sites, potentially using population-based weighting factors, will typically provide a time series that is a better representation of the exposure variations experienced by the population.

It is important to consider how long of a time series will be needed when a new set of measurements are initiated in support of acute effects studies. A rule of thumb is that 2–3 yr of daily air pollutant measurements for a population of about 1 million will provide sufficient power for reasonably precise determination of the magnitude of an acute effect on total nonaccidental mortality. Obviously, the more data the better, as shown in Volume 2 of the 2004 U.S. EPA Criteria Document for Particulate Matter (PM). The precision of the risk estimates tends to increase as the product of the daily death rate and number of PM measurement days increases (U.S. EPA, 2004). Constructing a time series of data with values every day is not an issue when hourly measurements are available. When integrated samples collected over 24-h periods (daily samples) are all that is possible (e.g., for measurements of some PM chemical constituents) then common practice is to collect samples less frequently than every day. For many years total suspended particulates (TSP) and $PM_{10}$ samples were only collected every sixth day in most circumstances. Although an improvement, many new networks are only providing samples every third day, which means it will still take three times longer to have sufficient statistical power for an acute effects or time-series study.

*Judging Site Representativeness*

For the most part, comparisons of monitoring site data within cities have demonstrated that day-to-day variations in outdoor pollutant concentrations (i.e., the acute exposure signal) are reasonably well correlated across a city. However,

this varies by pollutant and needs to be assessed before the data are used in time-series studies (i.e., acute health effect studies). For example, average inter-site correlations (i.e., for all available pairs of sites in Toronto in 2000–2003) for CO, $SO_2$, $NO_2$, $O_3$, $PM_{2.5}$, and $PM_{10}$ are 0.41, 0.78, 0.77, 0.90, 0.96, and 0.84, respectively. Similar correlations are present among stations in Vancouver and Montreal. The poorer correlation for CO compared to $NO_2$, both of which are predominantly vehicle-related pollutants in the city, can be explained by the uncertainty in the measurements as opposed to being due to more heterogeneity in concentrations. In Canada at least, improvements in the monitoring of CO or the treatment of the available data are needed before these should be considered in health studies.

For typical monitoring networks the pair of sites with the minimum correlation likely indicates a lower limit of the representativeness of using a single site to estimate population exposure. A low minimum correlation may also indicate that a given site is not appropriate by itself for use in a time-series study. Table 2 shows how the minimum correlation varied by pollutant and season within a smaller and larger region of Toronto. Not surprising, the representativeness of a given site's measurements decreased with distance and $PM_{2.5}$ varied most consistently across the region. However, the degree of correlation for $PM_{2.5}$ is not likely to be the case for the different PM chemical constituents. Three weeks of two per day sulfate, organic carbon, and elemental carbon measurements across three to seven separate sites in the downtown core of Toronto showed that sulfate was homogeneous while the carbonaceous constituents varied considerably (Brook et al., 2002). Organic carbon, for which there are many urban sources, from traffic to cooking, was the most variable. The minimum correlations in Table 2 reveal that a single site's $NO_2$ measurements have a greater potential to misrepresent the temporal variations experienced by the population. In Table 2 the lowest correlation between two sites was .08 ($NO_2$ in the warm season), suggesting that there is good potential for local conditions to influence the actual exposure experienced by members of the population residing further from the city center. A closer look at the site in question would be needed before a single measurement time series could be used for the larger Toronto domain.

**TABLE 2**

Minimum site to site correlation in daily average concentration variations across Toronto and outlying suburbs during 2000–2003. The Toronto correlations are for the group of sites within the core of the city (approximately a 20 × 20 km area). The metro area correlations include these sites plus sites in the suburban area (approximately a 40 × 40 km area). Warm corresponds to the warm season (May–September) with the cold season being October–April

| | | Toronto | Metro area |
|---|---|---|---|
| $PM_{2.5}$ | Annual | 0.91 | 0.90 |
| | Warm | 0.93 | 0.90 |
| | Cold | 0.92 | 0.86 |
| $NO_2$ | A | 0.56 | 0.29 |
| | W | 0.43 | 0.08 |
| | C | 0.67 | 0.40 |
| $SO_2$ | A | 0.62 | 0.43 |
| | W | 0.53 | 0.23 |
| | C | 0.77 | 0.42 |
| $O_3$ | A | 0.83 | 0.76 |
| | W | 0.77 | 0.69 |
| | C | 0.77 | 0.62 |

In general, the correlation in day-to-day variability across Toronto is reasonable for $NO_2$, $O_3$, $SO_2$, and $PM_{2.5}$. As suggested earlier, combining the time series from a group of sites can improve the representativeness of a time series. For example, for $NO_2$ the average and minimum correlation between the downtown Toronto site and the sites within the city are .83 and .75. These values increase to .89 and .86 when each of the sites is compared to the average time series generated by a combination of the Toronto sites. Similarly, when considering the suburban sites (i.e., those that are further away) the minimum correlation increases from .58 to .66 when the day-to-day time series is represented by the combined sites as opposed to just the central downtown site.

Simple examination of site-to-site correlations provides insight regarding representativeness of a time series and can help explain differences among pollutants in observed health effects from time-series studies or in measured personal exposures. However, it is difficult to specify criteria for a minimum correlation value, below which the sites and pollutants being considered would not be appropriate for deriving exposures to include in a time-series study. This issue requires further attention, including closer examination of site representativeness and the sensitivity of epidemiological analyses, particularly those attempting to consider multiple pollutants, to this source of exposure error. Logically, this error can be expected to attenuate the significance of true associations, but in multipollutant analyses there could be the potential to "transfer" the association to the pollutant that can be monitored at fixed sites with greater spatial representativeness and/or with a stronger link to actual personal exposures.

### *Links to Population and Personal Exposure*

Within limits, monitoring site locations can be optimized to best represent the potential exposure of the population. When combining information from sites the data from each site can also be weighted according to the size of the population within a certain radius. Measurements at sites with a greater surrounding population are thus counted more heavily. A potentially stronger link to exposure can be derived by considering

intake fraction (Marshall et al., 2003). This approach is more appropriately used for weighting the pollutant exposure risks posed by emissions from a variety of sources. Thus, assigning an intake fraction weighting to an ambient measurement also requires knowledge of the sources contributing to the observations and how the measurement, the source location, and the population of interest are related, including estimates of pollutant concentrations in a range of microenvironments. Probabilistic exposure models, such as the REHEX model (Winer et al., 1989; Fruin et al., 2001), also provide an approach for estimating, from ambient concentrations, the range of exposures expected among the population. Among other things, these models require more detailed spatial information on concentrations, either by interpolating the observations or through the use of air quality models.

Short-term field studies can help identify optimum site placement and the nature of the relationship with personal exposures. There have been several exposure studies examining how well outdoor measurements correspond to actual personal exposures. Brauer and Brook (1997) showed that day-to-day variations in outdoor central site $O_3$ measurements agreed reasonably well with the temporal variability in average personal exposures, but not necessarily for all people. In addition, the amount of agreement has been found to vary from city to city and among pollutants (Sarnat et al, 2001; Kim et al., 2006). The latter represents one source of differential exposure error.

$PM_{2.5}$ has been the primary focus of most recent exposure studies, and the results generally show that each individual's day-to-day personal exposure time series is correlated with the variations in the outdoor levels. However, there is wide variability from person to person depending upon their time activity, the predominant indoor microenvironments that they spend time in, and the indoor sources of $PM_{2.5}$ in these locations. Nonetheless, the median personal to ambient correlation among a sufficiently large group tends to be positive and strong enough to indicate that, on average, the exposure variations experienced by the population are captured by the monitoring site measurements. Attempts have also been made to separate personal $PM_{2.5}$ exposures into particles of indoor and outdoor origin (Wilson & Brauer, 2006). This has helped, for example, to assess whether there are differences in the impact on respiratory health of these two broad classes of $PM_{2.5}$ (Ebelt et al., 2005).

Ideally, when developing a time series for an acute effects study or quantifying the exposure gradient in a cross-sectional study (chronic exposure effects), the link to personal exposure should be tested. To date, there have been very few exposure studies geared toward comparing chronic personal exposure to the outdoor concentration "assigned" to that individual (Wheeler et al., 2008, Wheeler et al., personal communication), which is usually the concentration estimated at the person's home.

### *Limitations in Linking Air Quality Measurements and Health Data*

In addition to the exposure errors, it is also important to consider the nature of the health effects being studied and the type of health data that are or will be available. True acute responses likely arise immediately after exposure (i.e., within minutes to hours), but may linger and/or continue to grow in severity for at least a few days. This time course is not well understood, but can be expected to differ depending upon the physiological functions being affected (e.g., respiratory, cardiovascular), from person to person, and depending upon how the exposure conditions change during this period. Administrative health data do not provide the time resolution needed to improve understanding of this etiology. Emergency-room visits, hospital admissions, deaths, absenteeism, etc. are recorded as daily counts corresponding to midnight to midnight. While at a minimum this implies that only daily pollutant averages and exposure metrics derived from these values are necessary, hourly measurements can be utilized in the form of daily maximum readings over shorter durations. This permits some assessment as to whether the acute responses are associated more strongly with short-term peak levels or higher concentrations sustained over 24–72 h periods. Confounding among these different metrics will hinder definitive conclusions about what acute exposure period is of most importance to the population at large. However, it may be possible to gain some insight for more detailed follow-up using different study designs (e.g., prospective panel studies), and thus, hourly measurement data are valuable to obtain.

There are relatively few health and air quality databases for chronic exposure studies. Thus, when health data relevant for air pollutant effect studies are assembled, the exposure information is most often derived from whatever is available. In this case, prospective air quality measurements can be useful if they can be shown to correctly reflect the past and present gradient in exposure within the cohort. In the rare circumstance that a cohort is developed prospectively, it is more likely that air quality measurements can be included in the program (e.g., Spengler et al., 1996; Peters et al., 1999). Even so, demonstrating that such measurements are representative of longer term or even lifetime exposure remains an issue (Brook & Spengler, 1995), and mobility of cohort members may need to be taken into consideration (Jerrett et al., 2007a).

### *Improving Air Quality Information for Health Research*

More targeted air quality risk management will ultimately require more knowledge regarding the specific pollutants or sources posing the greatest risk to human health. Presumably, standards for these pollutants (e.g., a chemical component found on $PM_{2.5}$) could be established or control measures could be implemented on specific sources. Acquiring the necessary knowledge requires highly specialized health studies. These need to be supported with much more detailed ambient air pollutant measurements and a better understanding of the

relationship between ambient concentrations and actual exposures. These are likely to come from field studies as opposed to monitoring programs. The types of measurements needed and other health research-related needs that the atmospheric science community can potentially satisfy were recently discussed by NARSTO (www.narsto.org).

There have been considerable advances in measurement capabilities due to extensive research on $PM_{2.5}$ during the past decade. Applying these capabilities for health research can be expected to lead to new understanding (Wexler & Johnston, 2006). However, even stronger collaboration among health, exposure, and atmospheric scientists will be needed to take full advantage of these advances, given the level of understanding required to operate equipment and design effective studies. At the same time there is increasing interest in studying a wider range of health endpoints (e.g., cancer risks, impacts on the fetus and trans-generational effects) and a wider range of air pollutants (e.g., toxics, radon, aeroallergens). This will hopefully lead to a more holistic understanding and approach to risk management. However, achieving this level of understanding will be challenging, requiring a sustained multidisciplinary research effort in environmental health.

As monitoring programs are established or expanded, enhancing the spatial coverage of the sites could potentially improve the applicability of the data for future chronic health studies. However, even with close contact with health researchers it is difficult to anticipate all the areas that might need to be covered unless there are specific cohorts in mind. Even so, resource limitations usually make it impossible to monitor all locations, especially if only done on the speculation that a health study might use the information.

Intra-urban variations in exposure are now recognized as an important signal to exploit in chronic studies, and again, it is not likely possible to operate enough monitors. Mobile measurement platforms (e.g., Bukowiecki et al., 2002; Westerdahl et al., 2005; Yli-Tuomi et al., 2005; Kolb et al., 2004; Polina et al., 2004; Guo et al., 2006; Xu et al., 2006) can assist in studying spatial patterns and in optimizing site placement. They are becoming more common as a facility for monitoring agencies, and current advances in technology are allowing even more sophisticated measurements to be obtained. However, measurements alone cannot provide all the spatial detail desired. Thus, alternate sources of information and/or a range of spatial models, statistical or physical, are becoming increasingly important to develop. These are discussed later in fourth subsection.

### *Tracking Progress*

One of the primary objectives of monitoring networks is to track progress toward achieving standards and to insure that good air quality is maintained. Most mature networks have multiple sites that are in standardized locations, several if not all of the criteria pollutants are being monitored, and long data records have been and are continuing to be collected. Thus, they are well suited to satisfy this primary objective. Beyond selecting the right locations and pollutants to measure, best practice clearly dictates that for studying trends a long, unbroken record is important to maintain. This is because air quality changes can be small and gradual and are obscured by meteorological variability. Lack of continuity in measurements, lack of sensitive enough measurements, and underrepresentation in some geographic areas hinders trend analysis. Consequently, best practices also dictates that closing or moving monitoring sites with long records should be considered very carefully. When a site is moved, both the new and old site should be operated simultaneously for as long as possible in order to quantify concentration differences.

Demonstrating progress in direct response to implemented policies is now being referred to as “accountability.” Ultimately, this should extend beyond just documenting air quality improvements to demonstrating that the desired benefits have been realized. This would include, for example, improved public health, recovering ecosystems, and fewer poor visibility events. However, tracing back along the full accountability chain is very challenging (HEI, 2003), and the implications it might have for how air quality measurements are undertaken have not been fully resolved.

If accountability is limited to detecting the expected air quality improvements resulting from a new policy or a specific intervention, then time-series length may not be as important as for detecting gradual trends. However, sufficient, high-quality baseline measurements in advance of the emission reduction(s) are critical, as is continuation of these measurements after the reductions have been implemented. Existing sites providing the baseline data need to be identified and, if necessary, improved (e.g., adding high-sensitivity instruments or new measurements).

Air quality models and emissions inventories have improved significantly over the past 10–15 yr. This includes them being more widely applied (i.e., more accessible to a large number of users) and availability of sufficient computing resources for longer term simulations. Thus, there is an opportunity for models to play a greater role in informing monitoring activities. They can be used to simulate the magnitude of the change at the sites being used for tracking progress or demonstrating accountability. This may indicate that more sensitive measurements will be needed, given the anticipated changes, or that additional measurements at existing sites or at new locations would help to more effectively and more rapidly demonstrate accountability. Models can also be applied to identify the most useful measurement locations for “filling in” information between sites and, as discussed later, they can provide a more continuous picture of spatial patterns by merging their output with the measurement data. In some countries or jurisdictions, previous data and models have provided knowledge to allow downsizing of networks with limited or no loss of information (e.g., leaving sufficient sites to monitor trends and model population exposures), which helps reduce costs. Thus, given their current level of development, best practice clearly dictates

that models be used as much as possible to optimize and expand the usefulness of air quality measurement programs.

A weakness of many existing networks is that they have focused more extensively on urban areas. Consequently, there are much fewer rural measurements and the length of their time series is shorter. For example, worldwide, there are few long-term trends on rural $NO_x$ levels, despite its importance to understanding regional $O_3$ and $PM_{2.5}$ and the impact nitrogen deposition can have on ecosystems. Furthermore, this limits our current ability to assess how the growth in the size and density urban areas (e.g., sprawl) is impacting proximate regional air quality. Urban–rural pairs of measurements (Brook et al., 1999) are becoming more common. They are essential for untangling contributions and trends attributed to local/urban sources versus upwind sources from regional scale transport. Conversely, some classes of pollutants have traditionally been measured in rural and remote areas (e.g., persistent organic pollutants [POPs], acid deposition, sulfate), and thus, understanding of the conditions in populated areas is more limited.

More and more agencies responsible for air quality measurements are enhancing activities at selected sites (supersites or AQI sites or core sites) and potentially collecting fewer measurements at sites in between these locations. This approach supports a greater amount of air quality science at the supersites and can provide cost savings. The level of activity at the supersites often varies over time, depending upon funding and the current "hot issues." However, stability for a core set of measurement is important since these supersites are likely to be ideal for studying long-term trends and potentially for accountability purposes. The measurements obtained at the "in-between sites" or "satellite sites" typically depends upon the objectives of the program funding the work. With respect to air quality and smog, such sites are most likely to monitor $O_3$ and $PM_{2.5}$ or $PM_{10}$. This is driven by the fact that these pollutants are closer to or over existing standards and there is a perception that they are most important with respect to health effects. While this latter point is valid, in many areas these pollutants exhibit less spatial variability implying that fewer sites are needed. This is, perhaps, less likely for $PM_{2.5}$ because although its total mass may vary relatively slowly over space some chemical constituents, particularly those related to primary emissions, will potentially vary much more rapidly. Thus, more knowledge is needed to in order to determine how best to optimize the number of $PM_{2.5}$ sites and $PM_{2.5}$ speciation sites.

### *Modeling, Process Studies, and Source Apportionment*

Identifying specific management strategies requires a good conceptual model of the causes of the air quality problems. Both monitoring and detailed field study data are necessary to develop this conceptual model and to obtain a greater understanding of atmospheric processes. Spatial and temporal coverage is the greatest asset of monitoring network data. Combined with meteorological information (e.g., trajectories), relatively complete conceptual models can be devised and considerable information about contributing sources areas can be obtained (e.g., Brook et al., 2004b). As highlighted by discussion and examples in Chapter 6 of the NARSTO PM Assessment, all forms of data analysis, from the simplest to the most complex, will provide insight. However, this requires that resources be continually dedicated to this purpose and that experienced analysts, with backgrounds in atmospheric chemistry, meteorology, and statistics, are employed.

Monitoring network data can also play a valuable role in model evaluation. This role is becoming more important because many models are being run continuously for AQ forecasting and detailed field studies cannot be undertaken continuously. These new long-term modeled data sets are offering new opportunities to learn a great deal about how the models perform and the quality of the emissions information. While the network data tend to lend themselves more to operational evaluations of the model, there are opportunities for diagnostic evaluations as well (see second subsection). Increased use of continuous or semicontinuous particle composition instruments for monitoring can be expected to provide greater opportunities for such evaluations. Network data are also critical to define the model's initial conditions. Advances in rapid data assimilation (Ménard & Robichaud, 2005) have been occurring for this purpose. These need to be continued since there are many potential applications of these assimilated data sets.

More detailed measurements, which can only be sustained for relatively short field studies, are ultimately needed to study atmospheric processes (dynamical, chemical, and physical) and for more detailed diagnostic model evaluations. The scope of these studies can vary greatly from a small team collecting measurements to study one process (e.g., Makar et al., 1998) related to one model module to large collaborations across institutions (e.g., EMEFS, ICARTT; Dennis et al., 1993; Frost et al., 2006). Just as there have been advances in what can be reliably measured at monitoring sites, there have been significant gains in what can be measured during field studies. Methods have improved and some advanced technologies are commercially available and are reasonably reliable and straightforward to operate with highly qualified personnel. This means that the ease with which a field study with highly comprehensive and technical measurements can be launched is now much greater than 5–10 yr ago. This is beneficial for obtaining, quickly, much more data at more locations from which one can study, in detail, source apportionment and atmospheric processes. However, the risk has never been greater for valuable data to be underinterpreted. Best practice obviously dictates that this be avoided. Thus, with these new capabilities comes an increased need for highly qualified and creative experts in data analysis along with careful advance thinking regarding the underlying hypotheses motivating any venture into the field.

### *Public Information*

In the long run, informing the members of the public is critical to the process of air quality management. They are the

ultimate decision makers since when a large enough majority decide that an issue is important, elected officials are more likely to respond. This can create opportunity for progress on air quality. Science/environmental advisors must be ready with the right advice, based upon emissions, measurements, and modeling information, when government leaders are prepared to make decisions. In recent years, health research results have garnered considerable attention in the media and, as discussed earlier, air quality measurements are critical to this research.

In terms of public information, the Air Quality Index (AQI) has been utilized for many years. In cities or regions where there are more frequent bad air days the public is generally more aware of AQ issues, at least partly due to the publicity of the AQI. The form of the AQI is similar in many countries, reporting air quality using descriptive terms such as good, moderate, poor, very poor, unhealthy, etc. (www.msc-smc.ec.gc.ca/aq_smog/aqcurrent_e.cfm; http://airnow.gov/index.cfm?action=static.aqi). This approach is easier for the public to understand and act upon as opposed to reporting actual pollutant concentrations. It has been designed to identify the worst effects that may result from the mixture of pollutants currently being measured and to describe the prevailing air quality. However, ozone and particulate matter are more often the driving pollutants (i.e., leading to an AQI other than very good or good). The AirNow web site (http://airnow.gov/), which reports actual ozone and $PM_{2.5}$ concentrations all across the United States and Canada along with color codes indicating the AQI, represents a significant advance in the information being publicly provided. It allows for the spatial extent of elevated air pollutant levels to be visualized and for users to quickly compare their region to others. The systems developed to obtain and synthesize this information and present it in near real time have only recently become possible, and this infrastructure is also critical for improved AQ forecasts. Similar systems for providing real-time AQ data and/or AQI values exist for many countries (e.g., the Netherlands: www.lml.rivm.nl/data/smog/index.html; Mexico: www.ine.gob.mx/). These are continually evolving and being integrated into multinational systems.

The increase in knowledge regarding air pollutant health effects has been leading to growing interest in upgrading or modernizing the AQI. One such program is the Air Quality Health Index (AQHI) being developed in Canada. This is currently being pilot tested in the Province of British Columbia (http://www.airplaytoday.org/). The goal of this pilot is to introduce the AQHI to the public and gather feedback, especially from people who are sensitive to air pollution. The unique feature of the AQHI is that it is based upon recent epidemiological results from across Canada. In addition, it considers multiple air pollutants simultaneously and they all contribute to the index value in every case.

The AQI has typically been for reporting current conditions so that the public can respond immediately. In recent years, however, the capacity to predict future $O_3$ and $PM_{2.5}$ concentrations, using physical and/or statistical models, has improved. Thus, the public can be informed in advance. There are also efforts being planned to predict more of the pollutants considered in the AQI or AQHI (e.g., $SO_2$, $NO_2$). Consequently, air quality advisories are being supplemented with daily air quality forecasts (www.msc-smc.ec.gc.ca/aq_smog/aqforecasts_e.cfm; www.msc-smc.ec.gc.ca/ aq_smog/chronos_e.cfm), but the information is not usually widely distributed (e.g., radio, television) until high levels, warranting an advisory, are predicted. Thus, despite the value of daily information for susceptible individuals, it is unclear if it is influencing the general public. Daily forecasts are also available in the United States (www.arl.noaa.gov/ready/ozone/), where some media are publishing maps every day. Countries outside of North America are also providing forecasts (e.g., www.epa.vic.gov.au/Air/AAQFS/default.asp).

Regular air quality forecasts, made possible through real-time reporting and assimilation of measurements and advances in air quality modeling, clearly does present an additional opportunity, beyond advisories, to inform the public of air quality issues (i.e., increase public awareness). A telephone-based survey undertaken after the original air quality advisory program for Canada had started indicated there was partial success in achieving this goal (Stieb et al., 1996), but few actually changed their behavior. Ideally, when public information programs are planned there should be some collection of baseline data, as opposed to retrospectively initiating such activities. This should give a truer picture of how the public's awareness and/or behavior changed.

Ultimately, poor air quality situations need to be minimized through preventative measures (i.e., new policies on emissions or activities producing emissions), and increased public awareness helps create the climate for political action. However, providing routine, reliable, and understandable air quality information to susceptible members of the population allows them to reduce their own exposure. This should not be underestimated as an important component of air quality risk management. Therefore, communication plans and health messages require careful consideration and regular evaluation for effectiveness.

### *Technical Issues in Establishing a Measurement Program*

Air pollutant measurements should only be taken if there is an ongoing commitment to a recognized standard of quality, and a plan for data archival and for interpretation. Poor quality and/or incomplete data or data of unknown quality have limited usefulness. It must also be recognized that knowledge and technology are continually improving and thus, to the extent possible, new measurements should seek to use the most current, accepted methods. This will increase the probability that the data are acceptable far into the future. In the long run, a small amount of high-quality measurements will be of more value than many measurements collected with insufficient documentation, quality assurance and interpretation.

Air quality measurement activities generally fit into one of two categories:

1. Monitoring—A core set of systematic measurements at well-selected locations that are maintained indefinitely for trend analyses (i.e., evaluate effectiveness of current policies), to determine whether an area is complying with or achieving an official air quality standard or guideline and to identify emerging problems as soon as possible, which may involve ongoing environmental health studies (e.g., epidemiological studies).
2. Field studies—A relatively short period (<2 yr) of more detailed or more specific measurements collected within a well-defined geographic area or at a given location or for a given population. These data are essential for development of conceptual models, source-oriented models, more-refined source apportionment studies, and for understanding the relationship between emissions, ambient concentrations and personal and/or population exposure. Various prospective health studies may also derive their exposure information from air quality field studies.

Data from monitoring programs lend themselves to a standard set of reports documenting current conditions, trends and comparison with other geographic areas. Quick release of such information into the "right hands" helps to keep air quality management issues in the forefront. A range of user-friendly software tools that can process the air quality data along with meteorological data is becoming common. This enables air quality scientists and managers to examine some of the causes of pollution events in near real time, providing information that in the past could take a year or more to obtain.

Field study data are usually more complex and less standardized. Sufficient time and resources need to be dedicated to working with the data after the study. At a minimum, 1–2 yr is likely necessary. To guide the planning of the study and subsequent data analysis there needs to be a set of testable hypotheses in place well before the study begins. Generation of new scientific knowledge is likely part of the study objectives and thus the first official reporting of the data tends to be in peer-reviewed journals. This is the only approach for ensuring credibility of the data, which is ultimately necessary for air quality managers to be confident with subsequent policy decisions. Prior to journal publication, preliminary results often appear at conferences, and before that, formal workshops help maintain momentum and offer an opportunity to combine data and build consensus. Air quality managers or their advisors should participate at this stage in order to stress the policy issues they are expecting the results will help inform.

Air quality measurement programs are expensive and so, in advance, must have clear short-and long-term objectives. Ideally, the program will be sufficiently flexible and broad to support multiple objectives, some foreseen and others not yet appreciated. Additional resources/expenses to insure data completeness, quality, analysis, interpretation, and reporting should not be overlooked since they are likely to be incremental (i.e., a small cost relative to the overall cost of obtaining and maintaining the data). The key technical issues to consider when establishing a measurement program are:

What to measure and how often:

- Ideally, multiple pollutants should be measured at the same site to assist in interpretation and to serve more than one objective.
- Temporal resolution; could range from seconds to days.
- Measurement methods to be utilized.
- Length of time series to be collected.
- Personnel needs in the field and lab/office.
- Criteria for introduction of new technology or additional pollutant measurements if the measurements are part of a longer term program.

Siting criteria and where to measure:

- Impact of local sources—may be desired or important to avoid.
- Type of sites include: source-oriented, such as curbside or other high-impact areas (e.g., hot spots); local or neighborhood scale; urban background; regional background (indicative of long-range transport).
- Representativeness to population and/or to region needs to be assessed.
- Geographic coverage and spatial density of sites if the program involves a network.
- Site access, serviceability, and security.
- Documentation of site meta-data.

Quality assurance (QA):

- Required level of accuracy, precision, and data completeness.
- Frequency of collection of specific QA measures (e.g., duplicates, blanks, zeros, spans, calibrations, external audits).

Data archiving and reporting:

- Data turn-around time and policy for data exchange and criteria for permitting use in publications.
- Provision of data to national or international public archives.

Consistency with methods used at other sites in the same and different networks and between countries.

Addressing these technical issues, which are expanded upon later in this review, before measurements start helps ensure that the data are of greatest value. In establishing a measurement program it may also be relevant to consider the potential applicability of the data for evaluating models and also for integrating the data with model output and other information to improve the detail and coverage of ambient concentration information (see later discussion). Linkages to personal exposures and the ability to quantify the degree of exposure

error associated with using the measurements for health research may also be necessary to consider.

*What to Measure and How Often*

For a wide range of both gaseous and particulate pollutants, Chapter 5 of the assessment published by NARSTO (2004) provides considerable detail on what can be measured, the methods available, how reliable they are, and reasons such measurements might be needed (e.g., for health effects studies, compliance monitoring, visibility, scientific understanding, etc.).

When feasible, greater frequency of measurement (i.e., finer time resolution) is preferred because this permits a much better understanding of source contributions and atmospheric processes (Wexler & Johnston, 2006). If a standard exists, then its "form" or "metric" (e.g., hourly maximum, 8-h maximum, 24-h, annual) will dictate that a certain resolution be achieved. Inclusion in a real-time reporting program, such as may be needed for air quality index and air quality advisory purposes or for air quality forecasting, will also likely demand that data be available on a frequent basis (e.g., hourly). Choice of resolution also has an impact upon the resources needed for QA and for data storage, as well as for data analysis and interpretation activities. Ultimately, the time resolution that is measured is dictated by instrument capabilities.

Although air quality standards or other types of regulations/guidelines require that several common pollutants are monitored indefinitely, several other pollutants or trace gases are important to measure in support of air quality management. This wide range of trace atmospheric chemicals can be classified in a variety of ways. Here we choose to consider four classifications; however, with any such attempt, the distinctions are blurred. These classifications are:

- Pollutants formed during combustion.
- Pollutants released from the surface or fugitive releases.
- Volatile organic compounds (VOC).
- Secondary pollutants.

Table 3 provides a summary of pollutants under each category and highlights important requirements and considerations for their measurement. Arguably the largest group, in terms of quantity of emissions, contains *pollutants associated with combustion emissions*. Many of these pollutants are produced and emitted simultaneously, which presents opportunities for comanagement. This includes nitrogen oxides (NO, $NO_2$, or $NO_x$), carbon monoxide (CO), fine particles ($PM_{2.5}$), ultrafine particles ("ultrafines" or $PM_{0.1}$), and, depending upon the presence of sulfur in the fuel, sulfur dioxide ($SO_2$). The first two, plus $SO_2$ and some form of particles (e.g., total suspended particulate, TSP; particulate matter less than 10 μm in diameter, $PM_{10}$), are generally referred to as criteria pollutants.

Particles are monitored according to total mass below a specific size, but in the case of ultrafines, measurements are based upon total number. Combustion particles are also composed of many different chemical compounds, which are discussed later. However, one important particle-phase constituent that should be explicitly included in this group is black carbon (BC). Also referred to as elemental carbon or soot, BC is being measured more frequently due to its strong link to traffic emissions, especially diesel particulate matter, which is becoming more and more recognized as posing a risk to health. In Germany, the National Environment Agency has implemented a BC ambient concentration limit of 8 μg/m$^3$ (arithmetic annual average value), and regulations in other jurisdictions (e.g., California) are being considered or are in place).

Carbon dioxide ($CO_2$) is typically the most abundant trace gas in combustion emissions, thereby linking, at the source, the issues of air quality and climate change (note that BC and aerosols are also important pollutants regarding climate). All of the pollutants already mentioned can be measured with instruments that provide hourly or better time resolution and that are capable of reliable, real-time data storage and transmission. With the exception of the particle-related measures, regular, automated QA or calibration can be included in monitoring routines. Calibration of the particle instruments is generally not possible because absolute standards are not available. Other approaches to insure data quality are thus necessary.

Another relatively distinct class of pollutants of importance to measure can be characterized as being *associated with noncombustion surface or fugitive releases*. This class includes both those related to human activities and natural emissions. For many of these compounds current capabilities do not permit high frequency measurements. Agricultural emissions such as ammonia ($NH_3$) and methane ($CH_4$), as well as dust, pesticides (persistent organic pollutants, POPs), and bioaerosols, fit in this class. Releases from a range of industrial processes and waste management are also important sources. In addition to the pollutants just listed, reduced sulfur compounds (e.g., $H_2S$) best fit into this class of pollutants. Similarly, resuspended dust, which is typically in the coarse ($PM_{10\text{-}2.5}$ or $PM_{coarse}$) particle and giant particle (i.e., >$PM_{10}$) size ranges, best fits in this class. This includes road dust and wind-blown dust from agricultural practices and from certain types of industrial facilities (e.g., mining and smelting).

Very few of these pollutants are measured routinely due to the lack of reliable, cost-effective techniques, because there are no air quality standards that necessitate monitoring and/or because they are only a problem in localized areas. For example, total reduced sulfur compounds (TRS) or $H_2S$ can be serious issues in the vicinity of pulp mills, sour gas flaring, and certain agricultural practices. In addition to country-specific air quality standards, international agreements often necessitate some level of monitoring. For example, POPs are routinely measured at several locations in support of the Stockholm Convention (www.pops.int/).

With respect to current air quality risk management issues, ammonia ($NH_3$) and coarse particles ($PM_{2.5–10}$ or $PM_{coarse}$) are considered to be the most important to monitor or otherwise gain more information on their levels and spatial and temporal

**TABLE 3**
Air pollutant classes, measurement capabilities and other issues to consider

| Pollutant class | Examples | Measurement capabilities | Comments |
|---|---|---|---|
| Combustion Emissions | Nitrogen oxides ($NO_x$ or $NO_2$) | Can be measured with instruments that provide hourly or better time resolution | • Off the shelf use of most instrumentation will not correctly measure $NO_2$ due to interference from other forms of oxidized nitrogen such as nitric acid, particle nitrate and peroxyacetyle nitrate (PAN).<br>• Unknown inlet losses from some of these species leads to additional uncertainty<br>• Relative size of this interference increases further away from high NO emissions areas (e.g. large cities) and when atmosphere is more photochemically active (i.e. summertime)<br>• The nature of this interference should be understood before reporting and using $NO_2$ concentrations. |
| | Carbon monoxide (CO)<br>Sulphur dioxide ($SO_2$) | Can be measured with instruments that provide hourly or better time resolution | • Measuring the low concentrations present in many areas requires higher sensitivity instrumentation.<br>• Since these low levels are below standards network managers are tempted to stop measurements or pay less attention to the quality of the low concentration values. This greatly hinders the use of these data in studying atmospheric processes, source apportionment and health effects. |
| | Carbon dioxide ($CO_2$) | Can be measured with instruments that provide hourly or better time resolution | • Links climate issue (i.e., greenhouse gas emissions) and air quality issue |
| | Fine particles ($PM_{2.5}$) | Total mass can be measured with hourly or better resolution, but the techniques available have limitations.<br>Integrated sampling on pre-weighed filters is the most widely accepted approach, but is not without uncertainties.<br>Instrumentation now exists for automated, semi-continuous measurement of the main chemical constituents. | • A portion of $PM_{2.5}$ is semivolatile and this is the main cause of the measurement uncertainty<br>• Main contributors to $PM_{2.5}$ mass are organic carbon compounds (40–60%), sulphate (20–50%), nitrate (0–50%) and ammonium (5–15%).<br>• The latter three have been successfully measured in many locations and countries via filter sampling and laboratory analysis.<br>• Measurement of the carbonaceous material, which is typically separated into elemental (EC) and organic carbon (OC), is more uncertain. A significant difference exists among the different approaches and among different laboratories determining OC and EC<br>• Sample collection artifacts caused by semivolatile OC and relatively high blank filter concentrations for OC lead to considerable uncertainty. |

*(Continued)*

**TABLE 3**
*(Continued)*

| Pollutant class | Examples | Measurement capabilities | Comments |
|---|---|---|---|
| | Black carbon (BC) | Semi-continuous measurement approaches are commonly used.<br>The simplest measurement technique is based upon light attenuation once a filter collects sample over a pre-determined length of time.<br>British smoke, coefficient of haze and soiling index are predecessors to the present day instruments measuring BC via light absorption. | • When BC is measured from filter samples using thermal techniques it is typically referred to as EC and OC is often measured at the same time.<br>• The distinction between OC and EC is operationally defined and method specific.<br>• For BC or EC there is strong link to traffic emissions especially diesel PM |
| | Ultrafine particles (UFP) | Measurement is based upon the total number of particle counts (per cubic centimeter) for all sizes below 0.1 µm in diameter.<br>A condensation particle counter (CPC) operated with no specific size separation at the inlet essentially measures UFP because particles below 0.1 µm completely overwhelm the remaining counts above this size (i.e., for all the rest of the particles from 0.1 to ~100 µm)<br>One minute or better time resolution is possible | • The smallest particle size present varies between 0.005 µm (5 nanometers) and 0.02 µm (20 nanometers) and so it is important to know the smallest detectable size and corresponding count efficiency for each CPC utilized for UFP measurement.<br>• Electrostatic classifiers upstream from the CPC yields information on numbers of particles within many size ranges (i.e., particle size distribution).<br>• Typical systems (scanning mobility particle sizers – SMPS) can discriminate 32 or more size ranges or bins from 0.01 µm to 0.9 µm, providing a size distribution every 10 minutes or better. |
| Non-combustion surface or fugitive releases | A ammonia ($NH_3$) Methane ($CH_4$) | Continuous or semi-continuous methods for $NH_3$ are available. | • Ammonia plays a role in $PM_{2.5}$ formation therefore improved knowledge of its behaviour and emissions is needed for $PM_{2.5}$ management. |
| | Pesticides (persistent organic pollutants – POPs) | Low frequency filter and PUF based samplers are commonly used.<br>Daily samples are very rare. | • Monitoring requirements may be country-specific and/or specified in international agreements (e.g. Stockholm Convention requirement for POP monitoring). |
| | Resuspended dust ($PM_{10\text{-}2.5}$ or $>PM_{10}$) | $PM_{10\text{-}2.5}$ is traditionally included as part of $PM_{10}$ or TSP measurements.<br>Filter-based technique are typically used, but continuous $PM_{2.5}$ instruments can be adapted for $PM_{10\text{-}2.5}$ | • Data limitations hinder determination of the health risk posed by $PM_{10\text{-}2.5}$ and feasibility of managing ambient concentrations<br>• Sources include road dust, wind blown dust from agricultural practices and from industrial facilities (e.g. mining and smelting), bioaerosols (e.g., pollen, spores) |

| | | | |
|---|---|---|---|
| | Total reduced sulphur compounds (TRS) or $H_2S$ | Very few of these pollutants are measured routinely and the techniques are widely variable depending upon the compound | • Releases from industrial processes and waste management<br>• Examples of sources are pulp mills, sour gas flaring and certain agricultural practices<br>• Data are also limited due to a lack of regulatory requirements for monitoring and/or the localized nature of the emissions |
| Secondary pollutants | $O_3$ | Can be measured with instruments that provide hourly or better time resolution | |
| | ~60–70% of $PM_{2.5}$ | Instrumentation now exists for automated, semi-continuous measurement of the main chemical constituents. | • Sulfate, nitrate and ammonium is essentially all secondary.<br>• Secondary organic aerosols varies in amount from near zero to 50% or more of the OC. |
| | Some VOCs and SVOCs | Semi-continuous to nearly continuous measurements are possible for some compounds using research grade measurement methods.<br>Integrated samples with canisters or traps are common. | • Even the more routine approaches for measurement for compounds such as formaldehyde and PAN require highly trained operators and/or capable analytical laboratories. |
| | Other oxidants (e.g., $H_2O_2$, OH) | Research grade measurement methods are possible | • Measurement can be very important for understanding atmospheric chemistry.<br>• They are only undertaken during detailed research studies.<br>• Measurement is difficult as methods are expensive, experimental and challenging to implement and some of the compounds of interest are very short-lived in the atmosphere. |
| Volatile and semi-volatile organic compounds (VOCs and SVOCs) | 100s of individual compounds<br>Examples:<br>Benzene<br>Tolulene<br>Xylene<br>1,3, Butadiene<br>Isoprene | Total non-methane hydrocarbons (NMHC) can be measured with hourly or better resolution.<br>Semi-continuous measurement methods are possible for some individual VOC and SVOC compounds.<br>Air samples can be collected in the field and analyzed with a good degree of accuracy and precision for the lower molecular weight compounds (i.e., fewer than about 10 carbon atoms).<br>Larger molecules and the more-oxygenated species are more difficult to measure with confidence.<br>Sorbant traps are more commonly used. | NMHC provides limited information since the compound(s) responsible for the higher concentrations (i.e., the dominant fraction of the total VOC) and/or the temporal variation will vary and cannot be discerned. In addition, inlet losses, which are not necessarily the same for all the VOC and SVOC species, adds uncertainty to the measurement.<br>VOC and SVOC measurement is a labour-intensive and expensive.<br>In many cases the dominant species are not those of greatest interest from the standpoint of either atmospheric chemistry (i.e., $O_3$ or $PM_{2.5}$ formation) or toxicity.<br>Commercially-available systems for time-resolved, real-time measurement of some compounds require highly trained operators and/or very capable analytical laboratories |

variation. Ammonia plays a role in $PM_{2.5}$ formation, and thus, improved knowledge of its behavior and emissions is needed to manage $PM_{2.5}$. $PM_{coarse}$ has traditionally been included as part of $PM_{10}$ or TSP measurements. However, as the monitoring focus shifts to $PM_{2.5}$ the need to continue to manage the coarse particle fraction is becoming an independent issue. At present, data limitations hinder determination of the health risk posed by $PM_{coarse}$ and assessment of the feasibility of managing its level, especially given the range of natural and anthropogenic sources that contribute.

*Secondary compounds*, which are distinguished by the fact that they form in the atmosphere, represent an important class of air pollutants to measure. A large fraction of the chemical constituents found on $PM_{2.5}$ (fine particles) are secondary. This includes sulfate, nitrate, ammonium and some organic species. In the gas phase, ozone ($O_3$) is the most well-known and commonly measured pollutant. However, there are several other secondary oxidants, acids or VOCs of importance, either because of their potential to have health or environmental effects or because of their role in atmospheric chemistry. Examples are hydrogen peroxide, the hydroxyl radical, PAN, nitric acid, nitrous acid, hydrochloric acid, formic acid, acetic acid, formaldehyde, acetaldehyde, 1–3 butadiene and acrolein. Some of these may be emitted directly (i.e., primary pollutant), but atmospheric formation is likely the most important source.

Although $O_3$, $PM_{2.5}$, and some specific secondary constituents of $PM_{2.5}$ (e.g., sulfate and nitrate) can be measured with relative ease, measurement is much more difficult for most of the other secondary compounds. The methods available are expensive, experimental, and challenging to implement, and some of the compounds of interest are very short-lived in the atmosphere. Yet their measurement can be very important for understanding atmospheric chemistry. Consequently, when possible, measurements are undertaken during detailed research studies. Even the more routine approaches for measurement, such as exist for compounds such as PAN and formaldehyde, require highly trained operators and/or very capable analytical laboratories.

*Volatile and semivolatile organic compounds* (VOCs and SVOCs) are associated with both combustion and fugitive emissions as well as secondary formation and are thus part of all of the three classes already described. SVOCs are also found in both gas and particle phase, depending upon their properties and ambient conditions. However, due to the large number of compounds, their complexity, the challenging nature of their measurements, and the tendency for many to be toxic, VOCs and SVOCs are considered here to be a separate class of air pollutants.

Total gas-phase nonmethane hydrocarbons are often measured with relatively simple instrumentation at routine monitoring sites. However, such measurements provide a limited amount of information since the compound(s) responsible for the higher concentrations (i.e., the dominant fraction of the total VOC) and/or the temporal variation will vary and cannot be discerned. In many cases the dominant species are not those of greatest interest from the standpoint of either atmospheric chemistry (i.e., $O_3$ or $PM_{2.5}$ formation) or toxicity. In addition, inlet losses, which are not necessarily the same for all the VOC and SVOC species, add uncertainty to the measurement. It is important to distinguish between $CH_4$ and the remaining VOCs because the former is often found in much higher concentrations and behaves differently in the atmosphere (e.g., has a much longer lifetime).

Measurement of individual VOC and SVOC compounds is necessary to provide insight into their contributions to $O_3$ and $PM_{2.5}$ formation. In addition, they are useful for source apportionment and in order to characterize ‘hot spots’ of high exposure to toxics. Air samples can be collected in the field and analyzed with a good degree of accuracy and precision for the lower molecular weight compounds (i.e., fewer than about 10 carbon atoms). Larger molecules and the more highly oxygenated species are more difficult to measure with confidence. In all cases, VOC and SVOC measurement is labor-intensive and expensive. Although there are commercially available systems capable of time-resolved, real-time measurement of some compounds, these require highly trained operators and/or very capable analytical laboratories, and careful consideration of the uses of the data and their subsequent storage is necessary.

*Particle composition.* A large amount of information on $PM_{2.5}$ mass measurement and $PM_{2.5}$ sampling and chemical analysis is provided by Chow (1995). Since that publication there have been significant advances in semicontinuous measurement of nitrate and sulfate, OC, and EC. These measurements, although more challenging than semicontinuous mass measurement and the traditional approach of using filters and laboratory methods, are providing new insights in the sources of $PM_{2.5}$. Some discussion on the various technologies being developed and applied is included in the NARSTO Assessment (NARSTO, 2004), and an overview of some of the new insights these instruments have enabled can be found in Wexler et al. (2006).

Organic carbon is one of the most challenging aspects of $PM_{2.5}$ measurement. As indicated earlier, sampling leads to uncertainties. The other main difficulty is that not all of the specific chemical compounds contributing to the total OC are known. In general, only about 20% of the OC can be consistently identified. These are chemical species such as polycyclic aromatic hydrocarbons (PAHs), alkanes, and a large variety of organic acids. There is a considerable amount of ongoing research on the chemical speciation of specific organic compounds and much more data, although far from routine, now exist. There is a growing body of evidence of particle surface chemistry contributing to the uptake of organic mass and altering the chemical nature of the compounds present. Such processes need to be understood much better, as do the emissions of OC and the contribution for natural sources (primary and secondary OC) before OC can be understood sufficiently to model specific control strategies. Thus, organic material on $PM_{2.5}$ currently represents

one of the greatest and most important challenges to the scientific community.

*Siting Criteria and Where to Measure*

Location of measurement and the immediate surroundings have a large impact upon the concentrations observed. The monitoring objectives play a large part in dictating the types of locations that are desirable. This typically leads to more than one network being necessary to support all issues. The U.S. EPA describes four categories of networks and sites and lays out the general purposes of the overall program and of each network on its web site: www.epa.gov/air/oaqps/qa/monprog.html. Similarly, a range of site categories, their purposes, and criteria for inclusion in EuroAirNet are described at http://air-climate.eionet.europa.eu/databases/EuroAirnet/euroairnet_criteria.html.

Standardization is important to consider for monitoring networks, and this becomes more challenging when they span multiple countries. Clearly, strict siting criteria need to be adhered to. For example, criteria for particulate monitoring in British Columbia are detailed at www.env.gov.bc.ca/air/particulates/amgv1pnc.html.

On a broad geographic scale, sites can be classified as either being remote, regional, urban background, urban exposure hot spot, or industrial. The spatial scale or "footprint" that each type of site can represent clearly varies, from being nearly continental for remote sites to less than neighborhood scale for industrial sites. Kuhlbusch et al. (2004) show a breakdown, by country, of what type of station (traffic, industrial or background) monitoring sites contributing to the European AirBase data set fall into. There is considerable variability among the 21 countries examined in the distribution of station type and with respect to what fraction is urban, suburban, or rural. Sites supporting air quality risk management typically fall within the urban exposure hot spot to regional scale. The latter type of site is best situated to provide information on transboundary transport as well as the general conditions in the rural area surrounding, but upwind of nearby urban areas (i.e., what is blowing into our cities from sources upwind). Ideally, such a site will be representative of at least a 20,000-km$^2$ area. The former type of site is best typified by measurements in high-traffic areas. These urban exposure hot spot sites should be representative of conditions that the general population is exposed to on a regular basis and/or of the conditions over a neighborhood experiencing higher concentrations due to the amount or type of emissions in the vicinity. Environmental justice issues are leading to an increase in the interest in studying high-exposure neighbourhoods. With the exception of industrial sites, the specific location of measurement should not be directly impacted upon by local emissions or by nearby obstructions to wind flow. The definition of local varies from the urban background to the regional sites, ranging from about 2 km to 25 km, respectively. Obstructions could be adjacent buildings in an urban area or the edge of a forest clearing at a rural or regional site.

The height of measurement is also an important siting criteria. Closer to the surface the measurements can be impacted upon by local dry deposition or local surface emissions or resuspension of dust. At regional and remote locations, 10 m is often the standard height as long as there are no nearby (~50 m) obstructions. In urban areas the concept of breathing zone enters into consideration, but there is a much greater risk that a very local emission will influence such a low measurement. Furthermore, low nighttime mixing heights tend to enhance this effect. Breathing zone measurements may be desired for specific exposure field studies, but are less likely to be representative of the urban background and are not likely to be representative of the conditions occurring within a hot spot. Alternatively, in cities, higher measurement heights such as rooftop locations tend to provide a better indication of the urban background and its temporal variation (Brook et al., 1999). Rooftops offer additional security for the equipment because they are difficult to access. This raises a key point about finding secure, accessible sites with power, communications, and a high likelihood of long-term stability, especially in urban areas and given budget limitations. The point is that compromise is sometimes necessary when selecting a site because few locations are "perfect" and access to a reasonable location may be difficult to find, given time constraints.

Existing measurements and/or past experiences can provide considerable insights in selecting measurement sites or designing networks. Air quality models can provide guidance and are also valuable for data analysis and for expanding spatial detail (i.e., conditions between sites). As many of these sources of information as possible should be used to optimize and enhance sampling strategies.

*Quality Assurance*

Documentation on the quality assurance (QA) measures and expectations that are or will be followed throughout a measurement program helps ensure the data's value and that they are not misinterpreted by other users. It is entirely the responsibility of the data generator(s) to initiate and uphold this plan and to make users of the data aware of the QA details. Providers of the funding for measurements should demand that evidence of a QA plan be available before a program proceeds. Conversely, QA details and other data limitations are important for data users to understand to ensure that correct conclusions are drawn.

Measurement methods and types and model numbers of the instrumentation used should be recorded, as well as any changes during the program. Details of the measurement site, such as latitude, longitude, elevation (above mean sea level, MSL), inlet height and design, and proximate emissions and obstructions (photos), are also necessary to document. This is referred to as site "meta-data." Most commercial instruments have known detection limits and levels of precision and accuracy as well as information on interferences. Nonetheless,

precision and accuracy targets for the measurements need to be quantified and the actual values being achieved should be determined routinely to ensure a failing piece of equipment is identified and replaced quickly to avoid data loss. Duplicate or repeated measurements of the same samples and analysis of standards with known, traceable concentrations are, therefore, critical to undertake routinely to track precision and accuracy, respectively. The amount of resources needed for a measurement program is impacted upon by the frequency of QA measurements (e.g., number of site visits, amount of standard gases used), but, as stated earlier, this aspect should not be underappreciated. On the other hand, when QA measurements are being made (e.g., "zero" or "span" readings), actual measurements are being missed, and thus the appropriate balance needs to be established.

AQ measurements that involve sample collection in the field followed by chemical or gravimetric analysis in the laboratory require QA in both the field and the lab. With respect to the field, one of the most important QA measures is the collection of field blanks. Whatever the approach to capturing the sample (e.g., filters, denuders, canisters, passive sorbants, traps or cartridges), at least 10% of the samples analyzed in the lab should be field blanks. These procedures should be clearly stated in the QA plan, and the resulting data need to be rigorously analyzed to ensure data quality objectives are upheld and/or adjusted.

QA measurements should be based upon concentration levels that are typical for the site and that are within the measurement range of interest to the program. QA samples should also be introduced into the instrument or the analytical procedure (i.e., for laboratory analysis) in a manner that mimics the real measurement process and the conditions during measurement as much as possible. In general, the more experimental the measurement method the more challenging the QA and more frequent QA checks are usually necessary. As discussed in the NARSTO Assessment (NARSTO, 2004), for some measurements appropriate standards are not available. In this case, method intercomparison can provide information to judge the level of confidence in the data. With respect to current AQ health issues of concern, BC and ultrafine particle (UFP) measurements are hindered by the lack of standards.

### *Data Archiving and Reporting*

Long-term storage of final measurement data, including the QA data and site meta-data, is of utmost importance. At the same time, data accessibility throughout the future needs to be simplified and rules for providing the data to all users also need to be considered. Whenever possible, data should be archived at their native temporal resolution since averaging to longer time periods will likely lead to unrecoverable loss of information that may be of value in the future. Given the present costs of storage media (hard drives, DVDs, etc.), archiving one-minute data is no longer an issue.

A key issue with AQ measurement programs is "data turn-around time." Objectives for this and rules for how other users can or cannot report the data need to be established. Faster turn-around can be expected to increase the value of the data, assuming that the more current the information, the greater the number of interested users and the greater the impact of publicizing what is being observed. There is an increasing demand for real-time data and data products reported via web sites and the media (discussed later). As indicated earlier, this is possible for a growing number of pollutants because of improved instrumentation, automated QA, and communications. Continued improvements in these areas can be expected to reduce subsequent QA work and to increase data usage and publicity.

There are a number of recognized national or international data archives or portals for data access. Each has its own criteria for accepting and then documenting and preparing or formatting the data for storage and exchange. Examples of national archives of standard monitoring data are the National Air Pollutant Surveillance Network (NAPS) maintained by Environment Canada's Environmental Technology Centre (www.etcentre.org/NAPS/ index_e.html), the Air Quality System (AQS), which is the U.S. EPA's repository of ambient air quality data (www.epa.gov/ttn/airs/airsaqs/), Instituto Nacional de Ecología's archive of Mexico's air quality data (www.ine.gob.mx/), and the Air Quality Archive (AQA) for data for the United Kingdom (www.airquality.co.uk/ archive/ data_and_statistics_home.php). Data from many European countries are also available from AirBase, which is under the European Topic Centre on Air and Climate Change (http://etc-acc.eionet.europa.eu/ databases/airbase/airbasexml/index_html).

A growing amount of North American AQ research data is being kept in the NARSTO archive. These data are available from the NASA Langley Atmospheric Science Data Center as ASCII data files, most of which are in the NARSTO Data Exchange Standard (DES) format. This format is described on the NARSTO Quality Systems Science Center site (http://cdiac.ornl.gov/programs/NARSTO/). Another common format in which data are provided is "NASA Ames" (http://cloud1.arc.nasa.gov/solve/archiv/archive.tutorial.html). Similar to NARSTO, the header section of a file contains important meta-data, including instrument type, instrument name, data resolution, and units. The Joint Research Centre of the European Commission uses the NASA Ames Data Exchange Standard (http://airispra.jrc.it/Start.cfm). The netCDF (network Common Data Form) library also defines a machine-independent format for representing scientific data (www.unidata.ucar.edu/software/netcdf/). Together, the interface, library, and format support the creation, access, and sharing of scientific data.

### *Consistency*

Proper QA and data archiving significantly increase the likelihood that measurement data are consistent among countries.

Utilization of instrumentation that has been approved by national or international standards organizations (e.g., U.S. EPA reference or equivalent methods, NIST) also helps ensure consistency. This is critical for successful AQ management and thus, the more there can be international consensus on QA requirements and data storage and sharing protocols, the better. Clearly, individual needs of each country and the fact that resources for measurements within a country will, by and large, be provided by that country imply that there will be differences. The people implementing the measurement programs will differ among countries. Establishing a small number of internationally supported master stations where a wide range of ongoing measurement comparisons can be undertaken will provide valuable insight regarding consistency.

## Air Quality Modeling for Risk Management

### *Introduction*

As noted at the beginning of this section, a crucial component in understanding and managing atmospheric pollution is our ability to quantify the links between emissions of primary pollutants or precursors of secondary pollutants on the one hand, and ambient pollutant concentrations and other physiologically, environmentally, and optically important properties on the other. Air quality (AQ) models provide this capability. Such models consist of mathematical representations of the relevant physical and chemical atmospheric processes that are then solved by means of numerical algorithms to obtain pollutant concentration fields as functions of space and time for a given set of pollutant emissions and meteorological conditions (e.g., Peters et al., 1995; Seinfeld & Pandis, 1998; Jacobson, 1999; Russell & Dennis, 2000; Reid et al., 2003). Figure 10 shows a schematic of the atmospheric "process web," including the numerous links and interconnections between different atmospheric components that should be represented in an AQ model.

AQ models are also referred to by other names, including chemical transport models, long-range-transport models, emissions-based models, source-based models, source-oriented models, and source models. Depending on the particular set of atmospheric processes included in such models, they can be classified into various categories such as photochemical models, acid deposition models, and particulate-matter or aerosol models (e.g., Seigneur & Moran, 2004). All of these models, however, include a representation of some atmospheric *chemical* transformations along with representations of emissions, transport, diffusion, and removal processes. The inclusion of chemistry typically requires consideration of time scales ranging from fractions of seconds to days in order to account for many important chemical reactions, and hence AQ model domains need to extend at least several hundreds of kilometers in the horizontal and up to at least the middle of the troposphere in the vertical for compatibility with the transport that can occur during a multiple-day simulation. Models of air pollutants that do not consider chemistry, on the other hand, are generally referred to as "dispersion models."

Figure 11 shows a flow chart of the data flow required to apply an AQ model. In fact, as this figure makes clear, it is more accurate to refer to this as an AQ *modeling system* since the emissions files and meteorological files that are needed to drive an AQ model are provided by two other complex computer programs, namely, an emissions processing system (e.g., Dickson & Oliver, 1991; Houyoux et al., 2000) and a numerical weather prediction model (e.g., Seaman, 2000). An emissions processing

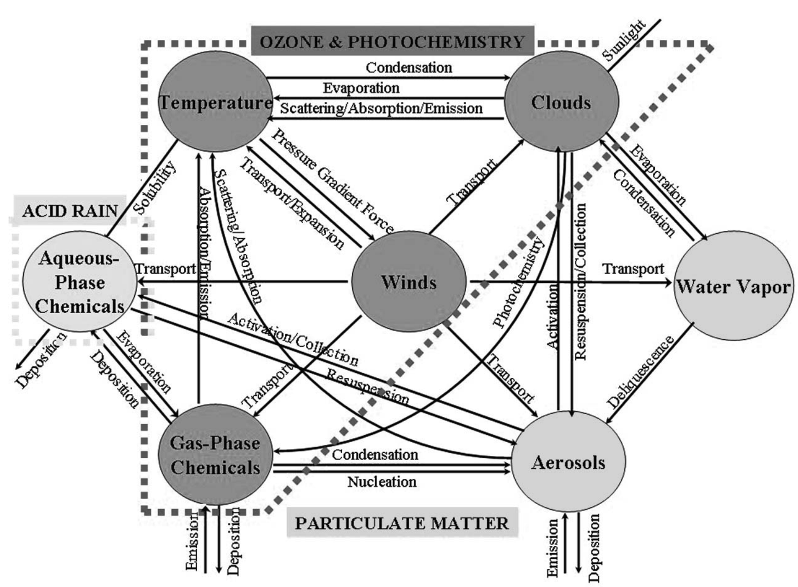



**FIG. 10.** Schematic diagram of atmospheric physical and chemical components and their interactions (adapted from Peters et al., 1995).

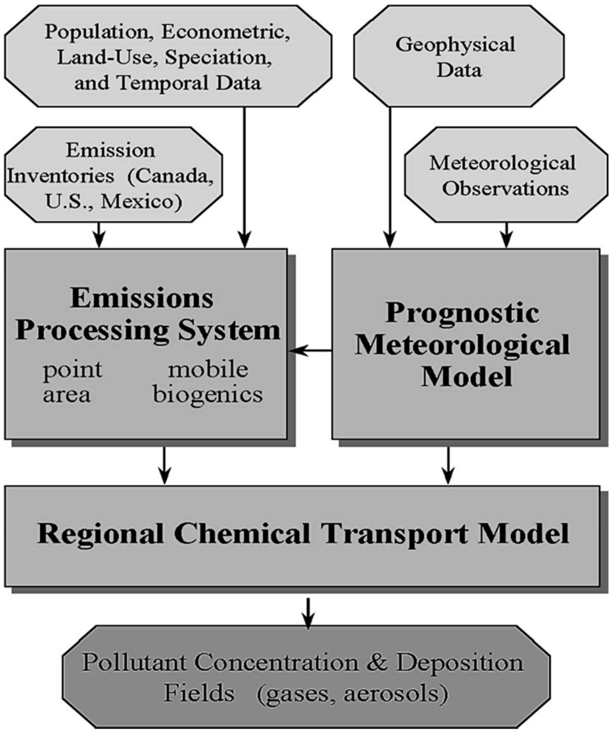



**FIG. 11.** Schematic description of the components of an AQ modeling system (Seigneur and Moran, 2004).

system in turn requires one or more emission inventories as its primary input plus such ancillary information as population, socioeconomic, and geophysical data. A numerical weather prediction model needs meteorological observations from a variety of observational platforms (e.g., surface instruments, rawinsondes, aircraft, satellites) as its primary inputs plus various geophysical data sets.

AQ modeling systems can be used to quantify source–receptor relationships for a wide range of air pollutants. They are also the only tool available to predict *future* air concentration and deposition patterns based on possible future emission levels. That is, they are prognostic in nature, unlike receptor models, which depend upon ambient measurements and hence are applicable only to periods for which measurements exist. For AQ models to be useful, however, there must already be information available about emissions and atmospheric measurements. If such data are not available for a region, then AQ model applications for that region can seldom provide much useful guidance for policymakers, due to the much greater uncertainties associated with model predictions due to the uncertainties in model inputs. (The need for emissions data is obvious as illustrated by Figure 11. AQ measurement data are needed to specify chemical initial conditions and boundary conditions as well as to evaluate model performance.)

AQ models have been used for decades to support AQ management, but at the same time they have also undergone continued and rapid development. For example, the first meeting in the long-running NATO-CCMS series of international technical meetings on air pollution modeling was held in Eindhoven, The Netherlands, in 1971. Most air-pollution models at that time were short-range, single-source dispersion models, and multiple-source models for modeling primary pollutants in urban settings were just being developed. Some of the earliest air-pollution models with parameterizations of chemistry were developed in the 1970s to simulate either the formation of photochemical smog in the Los Angeles basin or the long-range transport and transformation of air pollutants contributing to acid deposition in Europe and in North America. Given this considerable history, there have been a number of overviews of AQ models and AQ modeling over the years, including textbooks such as Jacobson (1999) and review articles such as Peters et al. (1995), Russell and Dennis (2000), Seigneur (2001), and Seigneur and Moran (2004).

The discussion in this section is not intended to provide a comprehensive review of AQ models and modeling practices. Instead, it builds upon the earlier NERAM paper on AQ modeling by Reid et al. (2003) and focuses on AQ model capabilities and uncertainties and on the management of these uncertainties in AQ model applications. This subsection (1) summarizes the variety of ways in which AQ models can contribute to AQ risk management; (2) reviews key technical choices and issues related to AQ model applications, especially those factors contributing to model uncertainty; and (3) provides an overview of "best practices" for using AQ models and their results and for managing associated uncertainties. Finally, conclusions and recommendations are presented.

*Applications of Models for AQ Risk Management*

AQ models can be applied in a number of ways, both directly and indirectly, to support AQ management and policy formulation:

- Evaluation of impact of emissions changes, including proposed control measures.
- Source apportionment and source attribution.
- Input to conceptual model development.
- Emission inventory evaluation.
- Measurement network and field experiment design.
- AQ forecasting.
- Testing current understanding of science.

Let us consider each of these applications in turn.

*Evaluation of impact of emissions changes.* The use of AQ models to assess the impact on air quality of emission changes due to pollutant abatement strategies, new pollution sources, population and economic growth, etc. has probably been the most common application of AQ models. Reid et al. (2003) gave three examples of such applications for Spain, Australia, and Canada, respectively: (a) the assessment of ozone abatement strategies for the Greater Madrid area (Palacios et al., 2002); (b) the impact of alternative urban forms of the city of Melbourne on urban air pollution levels (Manins et al., 1998); and (c) the

sensitivity of PM concentrations in Ontario to changes in emissions of primary PM and PM precursor gases. Four more examples include (d) the generation of intra-European source–receptor matrices ("blame matrices") by individual country for oxidized sulfur, oxidized nitrogen, and reduced nitrogen species for 2003 emissions and projected 2010 emissions (EMEP, 2005), (e) the assessment of the impact of possible emission control strategies in the Pearl River Delta region of southern China (Streets et al., 2006), (f) the estimation of AQ benefits from implementation of $SO_2$ and $NO_x$ emission reductions under the 2005 U.S. Clean Air Interstate Rule (U.S. EPA, 2005b), and (g) the evaluation of the potential impacts of proposed $SO_2$ and $NO_x$ emission control measures in Canada and in the United States on acid deposition in Canada (Moran, 2005).

Many of these studies follow a similar approach. The AQ model is first run for a "base-case" simulation, for which the emissions used are either historical or current and for which AQ measurements are available with which to evaluate model performance, and then again for one or more emission "scenarios," in which the assumed emissions correspond to a possible future state. For the simplest type of emission scenario, a "roll-back" scenario, emissions of one or more species may be changed by a fixed percentage for all source types across either the entire model domain or a selected subdomain. In more realistic emission scenarios, selected source types such as on-road motor vehicles or coal-fired electrical generating stations may be targeted. Sometimes one of the scenarios corresponds to a "business-as-usual" (BAU) scenario, in which emissions from the base case have been projected forward in time to account for population and economic growth and the implementation of any scheduled control measures expected by the scenario year. A companion future-year emission scenario may then be run that is identical to the BAU scenario except for the addition of a new candidate abatement strategy. Comparison of the base case, the BAU future scenario, and the companion future scenario then allows the impact of the abatement strategy to be estimated, as well as any changes expected in future-year AQ relative to current conditions if the abatement strategy were or were not implemented. As one example, Figure 12 shows predictions of effective acidity wet deposition from an acid deposition model for two cases: a 1989 base case and a realistic 2020 emissions scenario.

*Source apportionment and source attribution.* AQ models can be used to investigate and quantify source–receptor relationships in a manner that is complementary to the use of receptor models, that is, based on a source-oriented framework rather than on a receptor-oriented framework (Blanchard, 1999). The simplest approach is the so-called "zero-out" approach, in which emissions from a particular source sector (e.g., petroleum refineries) or from a particular jurisdiction (e.g., a city, a province/state, or a country) are set to zero while leaving emissions from other source sectors or jurisdictions unchanged. Predictions from this case can then be subtracted from predictions from a base run in which all emission sources are considered to estimate the impact of the targeted source sector or jurisdiction. Two more sophisticated approaches are "source tagging" and inverse methods. In the former, pollutant emissions from particular source sectors or geographic locations are tracked in the model as separate ("tagged") species (e.g., Kleinman, 1987; McHenry et al., 1992; Kleeman & Cass, 1999a, 1999b; Zhang et al., 2005). In the latter, the adjoint of the AQ model can be constructed, used to quantify the sensitivity of the model to emission inputs, and then combined with ambient measurements, or else initial attribution results can be refined based on the synthesis inversion technique (e.g., Uliasz, 1993; Pudykiewicz,

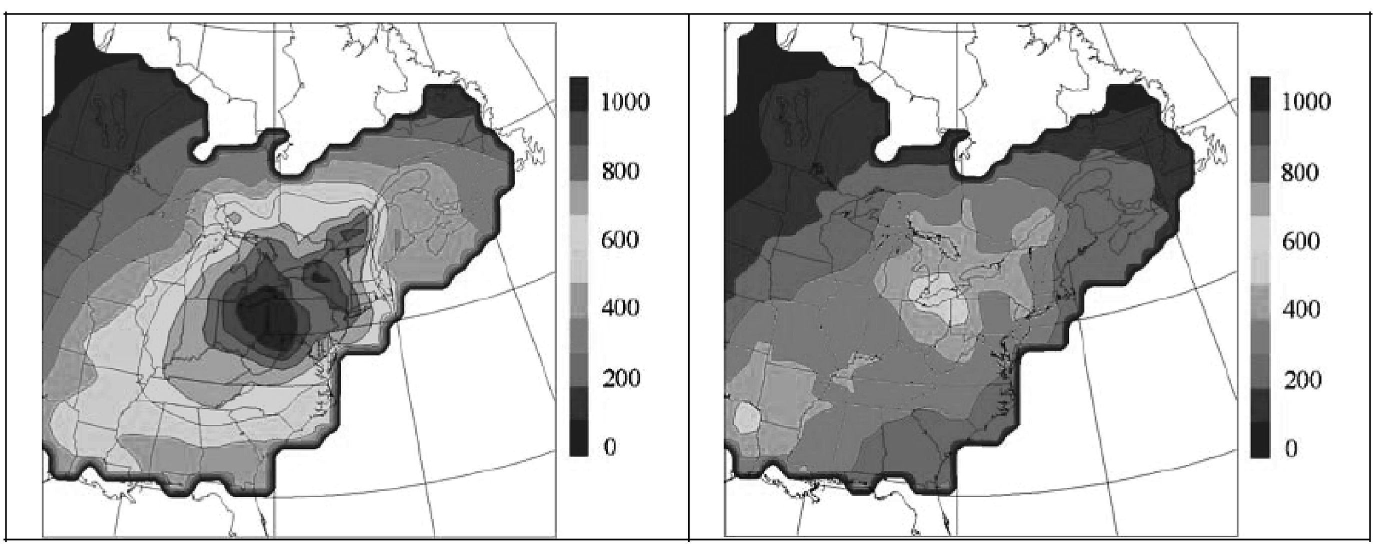



**FIG. 12.** Plots of annual effective acidity wet deposition (units of eq/ha/yr) predicted by the ADOM acid deposition model for (left) 1989 base case and (right) 2020 $SO_2$ and $NO_x$ emission scenario. Effective acidity is defined to be the sum of sulphate and nitrate wet deposition. See Moran (2005) for details.

1998; Mendoza-Dominguez & Russell, 2001; Mallet & Sportisse, 2005; Knipping et al., 2006).

*Input to conceptual model development.* An AQ conceptual model is a qualitative mental model for a geographic region that is based on a synthesis and simplification of available AQ information obtained from analysis of emissions, measurements, and AQ model results to distill the primary contributing factors, including key emission sources, terrain characteristics, and local weather and climate. Useful information from AQ models can include results from a suite of emissions scenarios, from source apportionment studies, and from sensitivity studies (see later discussion). One good example of the development of such a conceptual model is the analysis described by Pun and Seigneur (1999) for PM pollution in California's San Joaquin Valley. A number of the regional conceptual models for PM presented in the *2003 NARSTO PM Science Assessment* are based in part on AQ model results (NARSTO, 2004). One report (U.S. EPA, 2005c) gives a useful list of questions and points to consider in constructing a conceptual model of ozone pollution, including the relative contribution of local and distant sources, the role of certain weather patterns, and the nature of the local chemical regime. Answering some of these questions requires the application of AQ models. And in the paper by Zunckel et al. (2006), AQ model results contribute significantly to a conceptual model for surface ozone in southern Africa even with very limited availability of emissions data and AQ measurements.

*Emission inventory evaluation.* Since AQ model predictions depend directly on the input emission fields, the comparison of AQ model predictions with AQ measurements can give some indication of the accuracy of the input emissions. For example, if AQ model predictions are biased significantly high or low in a certain region as compared to measurements, one possible cause could be a corresponding high or low bias in the input emissions for that region. Or if measurements are stratified into rural and urban sites, then AQ model predictions for rural and urban locations can be checked independently to see if any systematic difference exists for rural vs. urban emissions (e.g., Yu et al., 2004). Inverse modeling analyses, in which enhanced AQ models are combined with ambient measurements, provide another, more quantitative, approach to estimating emissions strengths on either a regional or a global basis (e.g., Pudykiewicz, 1998; Mendoza-Dominguez & Russell, 2001; Martin et al., 2002; Palmer et al., 2003; Gilliland et al., 2006).

*Measurement network and field experiment design.* AQ models can also be employed as sophisticated interpolation schemes since their results are based on consistent and comprehensive representations of physical and chemical laws. For example, an analyst could use modeled pollutant fields as "true" patterns and then sample the model fields for different numbers of grid cells in order to investigate the impact of adding or removing stations to a network on the estimation of the actual pollutant spatial pattern from measurements, including the identification or nonidentification of strong gradients and "hot spots." Similarly, scientists planning and designing a field experiment could use model predictions to help them choose measurement site locations, aircraft flight tracks, and so on.

*AQ forecasting.* When AQ models are used to provide policy guidance or to interpret field-experiment measurements, historical periods are typically considered. Such applications are referred to as retrospective runs or "hindcasts." If, however, an AQ model can be run in "real time" quickly enough (i.e., in a few hours), then it can provide an AQ forecast for the next day or two, that is, a prospective simulation. Both the Canadian and the U.S. national weather services now run AQ models in order to issue public regional AQ forecasts (e.g., www.msc-smc.ec.gc.ca/aq_smog/chronos_e.cfm and www.nws.noaa.gov/aq/), and the Australian national weather service issues AQ forecasts for two large urban areas, Melbourne and Sydney (Cope et al., 2004). Besides providing useful guidance to the public about next-day AQ, such forecast programs have the added benefits of (a) maintaining or raising public awareness about AQ and (b) challenging the AQ models with a broader range of weather conditions than they are typically subjected to in scenario modeling (e.g., photochemical pollutant scenarios are almost always summer cases). Performance evaluations for AQ forecasts can then provide additional insights into model skill and reliability and identify model weaknesses (e.g., Eder et al., 2006).

*Testing current understanding of science.* Finally, AQ models provide a means to represent and link in a single package our best understanding of all of the chemical and physical processes relevant to AQ. This knowledge synthesis can then be evaluated by comparing model predictions with enhanced measurement data sets obtained from dedicated AQ field campaigns such as SCAQS (1987), EMEFS (1988–1990), NARE (1993), NARSTO-NE (1995), ESQUIF (1998–1999), BRAVO (1999), TexAQS (2000), ACE-ASIA (2001), ESCOMPTE (2001), Pacific 2001 (2001), TRACE-P (2001), and ICARTT (2004) (e.g., Dennis et al., 1993; Berkowitz et al., 1998; Heald et al., 2003, 2005; Frost et al., 2006; Hodzic et al., 2006; Pun et al., 2006; Smyth et al., 2006a). Conversely, an AQ model may also be used to help interpret field-campaign measurements, which can be difficult for a set of measurements of limited duration and restricted to a small number of locations due to the complexity of geography, meteorology, and interconnected chemical and physical processes. An AQ model can also be used as a test bed to test a new parameterization of a key chemical or physical process (e.g., Padro et al., 1993; Pierce et al., 1998). Such activities probe both our current scientific understanding and our representation of it in AQ models, often leading to improvements to both.

### *Key Technical Issues to Consider in AQ Modeling Programs*

Worldwide, there are a number of AQ modeling systems available, and each is typically composed of a set of large, complex computer programs. As a consequence, there are many choices to be made and issues to be considered by a modeler when using an AQ modeling system for any application.

Such choices and issues, however, also need to be taken into account by users of model results when judging the robustness and reliability of those results. Consider the following.

*Choice of model.* Some race-track devotees offer the advice that there are "different horses for different courses." The same is true of AQ models. The first step in applying an AQ model is to define the questions that need to be answered, and then, if possible, identify or develop an appropriate conceptual model. Doing so should immediately narrow down the set of AQ models that might be used to answer the question. For example, a very detailed but computationally expensive AQ model might not be the best choice for performing a multiyear AQ simulation, if such is called for. Another consideration is that a model designed to address one AQ issue (e.g., photochemical smog) may not include representations of all of the processes necessary to address another issue (e.g., deposition of acidic species—see Figure 10). And a model designed for highly polluted atmospheres may not be appropriate to model a clean atmosphere and vice versa (e.g., regional atmospheric chemistry in source regions vs. background global chemistry).

*Model configuration.* There are many choices to be made in configuring (i.e., setting up) an AQ model run. These include (a) the location and (b) the size, in both the horizontal and vertical, of the model domain, (c) the map projection to be used, (d) the grid spacing in both the horizontal and vertical, (e) the integration time step, (f) the simulation period, including any required "spin-up" time (the time for atmospheric concentration fields to reach an equilibrium between emissions and removal processes), (g) the "refresh" rate (the length of time that the *meteorological* model will be run before being re-initialized using a new set of meteorological analyses), (h) the set of chemical species to be considered, (i) the choice (in some cases) of parameterizations to be used for different chemical and physical processes, (j) the specification of initial chemical conditions, (k) the treatment of chemical lateral and upper boundary conditions, and so on. Each choice has implications. For example, the use of large horizontal grid spacing may "average out" a suspected hot spot or not represent small-scale meteorological circulations forced by local terrain features. See U.S. EPA (2005c) for a discussion about the choice of horizontal grid spacing, Berge et al. (2001) for a discussion about the specification of chemical initial conditions, and Brost (1988) for a discussion about the specification of chemical lateral boundary conditions.

*Science considerations.* One key limitation of AQ models is in fact gaps in our scientific understanding of the pollutants of interest. For example, it is well known that the sources of much of the carbonaceous component of atmospheric $PM_{2.5}$ are not presently known in spite of the fact that this component typically contributes ~40–50% of total $PM_{2.5}$ mass. Another example is our limited understanding of nighttime $NO_x$ chemistry (e.g., Brown et al., 2006). A third limitation is the use in current AQ models of process parameterizations of limited fidelity to the real atmosphere. For example, Dabberdt et al. (2004) recently identified the need for improved treatments of the influence on AQ of (1) the planetary boundary layer and (2) clouds and cloud processes. Another important consideration is the presence of nonlinear effects in the chemical reactions that generate some pollutants of interest. For example, the possibility of a nonlinear response in sulfate deposition to $SO_2$ emission reductions due to oxidant limitations was identified in the 1980s as a potential concern for managing acid deposition (e.g., Misra et al., 1989). Nonlinearities in ozone photochemistry are also well known (e.g., Seinfeld & Pandis, 1998), but PM chemistry possesses even more nonlinearities. For example, Meng et al. (1997) presented AQ model predictions for two simple ozone control scenarios run for a Los Angeles smog episode in which VOC emission reductions reduced ozone levels but caused increases in $PM_{2.5}$ mass. And West et al. (1999) presented AQ model results in which reductions in $SO_2$ emissions in eastern North America increased $PM_{2.5}$ concentrations due to so-called "nitrate substitution." Such nonlinear responses can further complicate the interpretation of model predictions and the formulation of possible emission control measures.

*Model parameterization and algorithmic limitations.* Even when AQ processes are well understood scientifically, they must still be represented mathematically in AQ models by so-called process parameterizations, and then the complex, coupled system of governing equations that comprise the AQ model must be solved numerically. Both steps have limitations and can introduce errors. For example, in many cases a number of different parameterizations have been developed to describe the same chemical or physical process, and these different parameterizations will produce different results (e.g., Kuhn et al., 1998; Zhang et al., 2000, 2001; Mallet & Sportisse, 2006). Typically, more sophisticated (and complex) parameterizations have a greater number of parameters and coefficients that must be specified, but measurements to do so may be scarce or lacking completely. This implies that while a more sophisticated scheme may have the potential to do a better job in describing a process, there are no guarantees that it will actually do a better job in practice. For example, some gas-phase chemistry mechanisms consider a few dozen species whereas others consider hundreds or even thousands of species. But besides reaction rates for all of the reactions that these species participate in, emissions must also be specified for each species, as must a number of chemical and physical properties, such as molecular diffusivity and Henry's Law constant, that are needed to estimate dry and wet removal rates. Such physicochemical data may not be available for every species (e.g., Zhang et al., 2002).

The numerical integration of the AQ model also introduces errors since it usually requires the solution of large coupled systems of both ordinary and partial differential equations. As discussed by Pielke (1984) and Jacobson (1999) among others, finite-difference methods are usually employed in both time and space. Any time-stepping scheme used to integrate the AQ model in time will have truncation errors that depend upon

both the order of the scheme and the chosen time step. Operator splitting is usually employed on the right-hand side of the governing equations to allow each process parameterization to be calculated separately, but operator splitting also introduces errors that depend upon the order of the splitting and the overall time step. Advection is well known to be a difficult process to solve, and literally hundreds of numerical schemes have been developed for advection. All suffer to varying degrees from some or all of truncation errors, numerical diffusion, phase errors, lack of positive definiteness, and violation of mass conservation.

Model "resolution" is another important consideration. The choice of a discrete model time step and grid-cell size implicitly imposes numerical filtering on the model solution. In essence, no temporal feature shorter than $2\Delta t$ and no spatial feature smaller than $2\Delta x$ can be predicted by the model, and $4\Delta t$ and $4\Delta x$ are probably a more realistic threshold (e.g., Pielke, 1984; Grasso, 2000). This has important implications for processes operating at smaller temporal and spatial scales (e.g., Pielke & Uliasz, 1998). As a consequence, many parameterization schemes to represent the influence of sub-grid-scale processes at grid scale have been developed. One obvious example is the representation of point-source emissions. In any Eulerian (grid) model, all or most point sources will be represented as volume sources since the emissions are assumed to be well mixed throughout at least one grid cell, thus introducing large numerical (i.e., artificial) diffusion in the vicinity of major point sources. To address this problem, which will be most pronounced for isolated sources, so-called plume-in-grid schemes have been developed to represent near-source diffusion and chemistry in plumes from major point sources more realistically. The treatment of vertical diffusion in any AQ model is also intrinsically a subgrid-scale parameterization since it must represent the impact of a spectrum of atmospheric motions that cannot be resolved by the model. Cumulus parameterizations that represent the impact of unresolved clouds on model fields are another important class of sub-grid-scale parameterizations that are employed in weather and AQ models (e.g., Haltiner & Williams, 1980).

*Input data.* AQ models require a number of input data sets in order to run. First are emission input files. Emission rates of a number of gaseous and particulate species must be specified for each model time step at each model grid cell at all levels. As discussed earlier, there are significant uncertainties associated with such emission files due both to errors in the emission inventories themselves and to additional uncertainties introduced by the emissions processing systems that perform the speciation and spatial and temporal disaggregation steps needed to create model-ready emission files (e.g., Hogrefe et al., 2003). For large point sources, ancillary information about smokestack characteristics such as stack height, stack diameter, stack-gas exit velocity, and stack-gas exit temperature is also needed. And if a future-year scenario is being considered, current emission inventories must be manipulated and modified to account for all assumptions built into the scenario.

Second, meteorological input files are needed if an "off-line" AQ model is being used (i.e., meteorological model and AQ model are separate), which is the case for most current AQ modeling systems. Meteorology is very important as it influences every aspect of the AQ system, including (a) natural sources of PM such as wind-blown dust and sea salt, (b) plume rise, (c) transport and diffusion, (d) gas-phase and heterogeneous-phase chemistry (via temperature and humidity effects), (e) cloud shading, (f) aqueous-phase chemistry, (g) dry removal, and (h) wet removal. Third, chemical initial conditions must be supplied for all model species for every grid cell, chemical upper boundary conditions must be specified for all model species at the top model level, and, for a limited-area AQ model, chemical lateral boundary conditions must also be specified for all model species. And fourth, a number of geophysical fields may also be required, including terrain height, land-use type, vegetation type, aerodynamic surface roughness, albedo, sea surface temperature, and soil texture. The accuracy and representativeness of all of these input files are a key concern, since even for a perfect model, the well-known aphorism "garbage in, garbage out" still holds.

*Infrastructure considerations.* As discussed by Reid et al. (2003), AQ modeling is typically resource-intensive in terms of model input data, people, calendar time, and computer power. In order to apply an AQ model for a particular case, the input data sets described earlier must be prepared for the model configuration selected, including emissions, meteorological, and geophysical files, the model must be run, and then the model results must be processed, analyzed, and interpreted. Typically, a minimum of three highly trained modelers would be required to contribute, namely, an emissions-processing specialist, a meteorological-modeling specialist, and an AQ-modeling specialist. The required calendar time from start to finish, including configuring and testing the model for the application, would likely be a minimum of weeks but more likely months. The minimum computer resources needed would be a high-end PC with multiple processors, large internal memory and disk space, and off-line archiving hardware to save numerous large model output files. Access to emissions data, meteorological data, geophysical data, and AQ measurement data is of course assumed as well.

*Model accuracy, sensitivity, and uncertainty.* For an AQ model's predictions to be used by policymakers, the model and its predictions should be credible. To be credible, the model should give the right answers for the right reasons. The determination of whether a model is giving the right answers is addressed by model performance evaluations, in which model predictions are compared to measurements. Model performance evaluation is discussed in more detail in the next section. However, there are some fundamental issues related to model accuracy, sensitivity, and uncertainty that need to be kept in mind. For one thing, how is the

"right answer" determined? For another, how can model uncertainty be determined?

There are actually a surprising number of issues that arise in comparing AQ model predictions with ambient measurements. The biggest one is *incommensurability*, which arises due to the fact that model predictions correspond to grid-volume averages whereas measurements are typically made at points in space or along lines (e.g., aircraft flight tracks, DIAL). For example, for a regional-scale AQ model whose smallest grid volume is 20 km by 20 km by 50 m, how representative would a single point measurement be of the 20 $km^3$ of air contained in that grid volume? One effort to address this question was made during the 1988–1990 EMEFS field experiment in eastern North America. Surface measurements were taken of 24-h $SO_2$, particulate (p-) $SO_4$, total (t-) $NO_3$, and hourly $O_3$ concentrations for 2 yr at 6 clusters of 3 to 5 stations that fell within the confines of model 80-km by 80-km horizontal grid cells. Cluster or subgrid daily variability was found to be approximately linearly related to mean concentration, with the largest variability associated with $SO_2$ and minimum $O_3$ (~ ± 70%), intermediate variability associated with p-$SO_4$ (~ ± 30%) and t-$NO_3$ (~ ± 40%), and the smallest variability associated with maximum $O_3$ (~ ± 20%) (Seilkop, 1995a, 1995b). Significantly, uncertainties due to this sub-grid-scale variability overwhelmed uncertainties associated with instrument error. McNair et al. (1996) performed a somewhat related analysis in the Los Angeles basin for $O_3$, $NO_2$, and CO for two 1987 SCAQS cases. For circular areas with 25-km radius, smaller than those considered during EMEFS, they found that local inhomogeneities for these three species had normalized gross errors in the 25–45% range.

A second issue related to the comparison of model predictions and ambient measurements is the need to compare "apples with apples." For example, for gas-phase species, AQ model predictions correspond to ambient conditions, whereas some networks report measurements at standard temperature and pressure (STP). For PM comparisons, model PM predictions are calculated based on Stokes diameter, whereas PM measurements are reported using aerodynamic diameter, PM measurements unlike models can suffer from artifacts related to volatile species such as nitrate, some organic compounds, and aerosol-bound water, and the distinction between elemental carbon (EC) and organic carbon (OC) is analysis method based and can vary from network to network (Seigneur & Moran, 2004). Note that the *definition* of the EC and OC variables predicted by an AQ model also depends on the EC–OC analysis method used to speciate primary PM emissions.

Turning to model uncertainty, the discussions and conclusions of a 1982 workshop on AQ model uncertainty have been described by Fox (1984) and Venkatram (1988). The latter identified three main sources of model uncertainty as "(1) errors in model inputs, (2) errors in model formulation, and (3) inherent uncertainty associated with the stochastic nature of turbulence." The last source constitutes a lower limit on model uncertainty since it cannot be reduced even if all model-related errors are corrected. One aspect of this inherent uncertainty is related to the time and space averaging used in measurements versus the ensemble averaging that is used to describe atmospheric turbulence. That is, atmospheric measurements correspond to samples from a single flow realization whereas AQ model parameterizations related to diffusion and mixing are based on ensemble averages for (theoretically) an infinite number of flow realizations with identical external conditions (e.g., Moran, 2000).

Reid et al. (2003) have noted that it is not possible to quantify overall model uncertainty because it is dependent on so many factors, some of them dependent on the particular application being considered, but also on the *interactions* of these factors. As already discussed, these contributing factors include errors and uncertainties in input data such as emissions, meteorology, and boundary conditions, uncertainties in our scientific understanding and in process parameterizations, errors associated with numerical methods, and uncertainties associated with required parameters like reaction rates. It is, however, possible to quantify some individual sources of uncertainty, particularly for numerical methods, to identify model sensitivity to various inputs and parameters, and finally to compare results from parameterizations and even entire models in order to try to characterize the range of uncertainty.

Error characterization is generally reported as part of the description of new numerical methods and parameterization techniques. A wide range of sensitivity analysis techniques exist, including DDM, ADIFOR, FAST, variational techniques, perturbation theory techniques, Green's function techniques, and stochastic techniques, that can be used to understand which model parameters and input variables most influence selected model outputs (see Zhang et al. [2005] for a useful literature review). Besides being compared side by side outside of models, the impact of different parameterization schemes can also be compared when embedded in a host model (e.g., Padro et al., 1993; Mallet & Sportisse, 2006). And some studies have compared differences in AQ modeling system results due to the use of different component models. For example, Hogrefe et al. (2003) compared the impact of using emissions files constructed by two different emissions processing systems from an identical emission inventory on the predictions of one AQ model. They found differences on the order of ± 20 ppb in predicted daily maximum 1-hour ozone concentration. Another source of uncertainty is meteorological inputs. Smyth et al. (2006b) compared the outputs from one emissions processing system and one regional PM model for two sets of meteorological input files for the same period that were provided by two different meteorological models. An operational evaluation of the two meteorological models suggested that their performance was essentially equivalent, as was the performance of the AQ model for the two sets of meteorological files, but when grid cells were matched for the same time, large variability was observed, particularly in aerosol quantities influenced by relative humidity. And recently, the performance of seven AQ models in predicting ozone was compared for the same period

(summer 2004) and region (eastern United States). The range of model predictions generally bracketed the measurements, and interestingly none of the models individually could match the skill of a weighted average of the seven forecasts (McKeen et al., 2005).

One other approach to assessing uncertainty is to synthesize expert opinion. Seigneur and Moran (2004) prepared a table that presented qualitative ratings of PM modeler's level of confidence in major aspects of the predictions of current PM AQ models. Only a few model aspects ($SO_2$, $NO_x$, and p-$SO_4$ air concentrations) were judged to have a "high" level of confidence. Most aspects were assigned "medium" or "low" ratings, and a few aspects, such as secondary OC and PM ultrafine mass and number concentrations, were assigned "very low" ratings. These ratings were based on an assessment of all contributing uncertainties, including uncertainties associated with the emissions of different pollutants and with scientific understanding.

*Review of Best Practice for Using Models for Air Quality Management* Let us now build upon the previous sections and consider that wry but wise epigram by Box (1979): "All models are wrong, but some are useful." That is, in applying models for AQ management, we must accept from the start that no model is perfect. Instead, as discussed in the previous section, AQ model predictions can be affected by numerous sources of error and uncertainty. How then can we account for the resulting uncertainty and apply models in a reasonable and defensible way in order to inform AQ management?

To start, how can we judge whether an AQ model will in fact be useful? For a model to be useful, presumably it must be *credible*. That is, it must have demonstrated sufficient skill and reliability that its predictions can be used with some confidence by analysts and policymakers in the formulation of AQ management strategies. Confidence can in turn be built in two ways: first, through *model verification* to assess the consistency, completeness, and correctness of the model and through *model performance evaluations* to characterize its performance and quantify its errors; and second, by applying the model in as appropriate, transparent, and defensible a manner as possible for the AQ issues being considered.

*Model verification and model evaluation.* Model verification and model performance evaluations should always be a required step before a model is applied in the policy arena. According to Fox (1981) and Russell and Dennis (2000), model verification is an assessment of the accuracy, reality, or truth of a model. It does not require a model to be run. Rather, model verification is a "desk check" in which the consistency, completeness, and correctness of a model's design, science, process representations, algorithms and numerical methods, inputs, and source code are examined and assessed. Peer reviewers should be involved in such an examination, and, ideally, any interested party should have unrestricted access to the model source code for this purpose.

Model performance evaluation is the process of examining and appraising model performance through comparison of model predictions with measured AQ data and/or predictions from other models (e.g., Fox, 1981; Dennis et al., 1990; Russell & Dennis, 2000). There are four main types of model performance evaluation: (1) operational, (2) diagnostic, (3) mechanistic, and (4) comparative (Seigneur & Moran, 2004).

An *operational* evaluation requires the statistical evaluation of model predictions of a few key pollutants of interest with atmospheric measurements over time and space scales consistent with the intended applications of the model. An operational evaluation is intended to answer the basic question: "Are we getting the right answers?" Examples of operational evaluations include EMEP (2003), Eder and Yu (2006), and Eder et al. (2006). A paper by Fox (1981) reviews a wide range of statistical measures that have been used in operational evaluations, but a recent U.S. EPA report (U.S. EPA, 2005c, Section 15.2) recommends a small number of statistical measures that have been found to be representative and useful in evaluating the performance of photochemical AQ models.

- A *diagnostic* evaluation is more of a research-level evaluation and involves an examination of model performance at the process level for all relevant species. A diagnostic evaluation addresses the basic question: "Are we getting the right answers for the right reasons?" Because diagnostic evaluations are more wide-ranging and comprehensive than operational evaluations and generally make use of nonroutine measurement data sets such as those from specialized field campaigns, they can identify the presence of compensating errors or excessive "tuning." Examples of diagnostic evaluations include Dennis et al. (1993), Karamchandani and Venkatram (1992), Sillman et al. (1998), Hogrefe et al. (2001a, 2001b) and Biswas et al. (2001), Heald et al. (2005), and Yu et al. (2005). Also, Seigneur et al. (2000) have described how to optimize the design of field studies that will be used in the evaluation of PM AQ models, a recent U.S. EPA report (U.S. EPA, 2005c, Section 15.3) lists some diagnostic analyses that have been found useful in assessing the ability of photochemical AQ models to predict changes in ozone due to changes in emissions of ozone precursors, and a recent paper by Zhang et al. (2005) examines three diagnostic probing tools that have been used to examine photochemical AQ model performance.
- A *mechanistic* evaluation involves testing individual model components (i.e., process representations) in isolation against field or laboratory measurement. Such evaluations address the question: "Are we using good parameterizations?" Some examples of mechanistic evaluations include Pleim and Xiu (1995), Odum et al. (1996), Geron et al. (1997), and Zhang et al. (2001).

- Finally, a *comparative* evaluation involves a side-by-side comparison with another model or model component for identical or similar inputs. A comparative evaluation addresses the basic question: "Are we getting comparable answers from comparable models?" Examples of comparative evaluations include Alapaty et al. (1997), Hass et al. (1997), Kuhn et al. (1998); Ansari and Pandis (1999), Zhang et al. (2000), Hogrefe et al. (2001a, 2001b), and McKeen et al. (2005).

Note that the term "model evaluation" denotes a process rather than an outcome or conclusion. For the terms "model verification" and "model validation," on the other hand, Oreskes et al. (1994) argued that numerical models of natural systems can never truly be verified or validated, since these terms imply the absolute correctness of a model. Fox (1981) and Russell and Dennis (2000) were careful to restrict their definitions of these terms. Model verification, as described earlier, refers to an examination process that at best leads to a provisional conclusion. And model validation is a process leading to a judgment on the quality, suitability, and usefulness of a model for a particular application that should be based on evidence from both model verification and multiple model performance evaluations. Such a judgment, however, must always be viewed as provisional, since additional information such as the results of a new evaluation may change the balance of evidence.

It is also important to consider which aspects of model performance need to be evaluated. Most AQ model evaluations involve case studies in which a model is run for a particular period using input emissions and meteorology suitable for that period and then model performance is examined using measurements from that same period. However, as already discussed, the most common AQ model application is to evaluate the impact of emissions *changes* on AQ. The key aspect of model performance in this instance is how well the model predicts the atmospheric *response* to the change in input emissions, and the approach to the corresponding performance evaluation is necessarily somewhat different. For a direct evaluation of model response, AQ measurements are required for *two* different periods so that an atmospheric response can be calculated, which means that the AQ model must be run for the same two periods using different input emissions corresponding to each of the two periods. Obviously, such a model-response evaluation is more demanding than the usual single-period evaluation since considerably more data and more modeling effort are required. Confounding issues include (1) the need to use emissions for two different periods estimated using a consistent methodology and (2) the additional variability introduced by interannual meteorological variability. As a consequence, published model-response evaluations are uncommon, but a few are available (e.g., Moran & Zheng, 2006).

Note that so-called "accountability" studies, in which the emissions changes that have occurred are due primarily to legislated control measures and the study goal is to assess the performance of the AQ models used to predict the benefits of those control measures *before* the control measures were enacted and implemented, are also model-response studies. Given the considerable time that will have elapsed, however, between the time the original AQ modeling runs were performed and the time that the AQ measurements were made following implementation of the control measures of interest, it is not likely that the particular version of the AQ model (or even the model itself!) is still being used. On the other hand, current AQ models can also be evaluated in a retrospective mode for the same legislated emission changes (e.g., 1985 Eastern Canada Acid Rain Program, 1990 U.S. Clean Air Act Amendments, 1998 U.S. $NO_x$ SIP Call). Note also that in terms of U.S. regulatory modeling terminology (e.g., U.S. EPA, 2001, 2005c), a model-response evaluation is equivalent to the evaluation of model-predicted *relative reduction factors*.

Finally, in considering the question "How accurate does a model need to be," Reid et al. (2003) suggested that the general answer is that "the model predictions should be good enough that model uncertainty does not affect the decisions that are based on the predictions." In the real world, of course, this may not always be the case. How then should models be used, given such uncertainties?

*Model applications.* In their review of photochemical models and modeling, Russell and Dennis (2000) discussed the *modeling process* as a separate topic. By this they meant the set of steps required to apply a model, including selection of model domain, grid resolution, and model configuration, preparation of model input files, model execution, and postprocessing and analysis of model predictions. The modeling process itself is worthy of individual attention because, as discussed in the previous section, all of these steps may influence the results provided by the modeling system. It is thus important to work through the modeling process in as reasonable and defensible a way as possible. Some limited guidance on how to do this does exist. For example, the U.S. EPA has prepared several documents to help modelers follow "best practice" when using regional AQ modeling systems for certain regulatory applications (U.S. EPA, 2001, 2005c).

Best practice basically boils down to thoughtful and careful selection, setup, and application of an AQ modeling system, accompanied by careful scrutiny and consistency checking of the results by various means, including the use of measurements and results from both alternate configurations of the selected AQ model and from other AQ models. Figure 13 describes eight steps of best practice for AQ modeling based on guidance from two U.S. EPA reports (U.S. EPA, 2001, 2005c). Most of the following steps will be relevant to any AQ model application.

Let us consider each step in turn. Some relevant background material has already been discussed.

1. *Formulate a conceptual model.*

   Both modeling specialists and modeling "clients" should have a conceptual understanding of the AQ issue to be

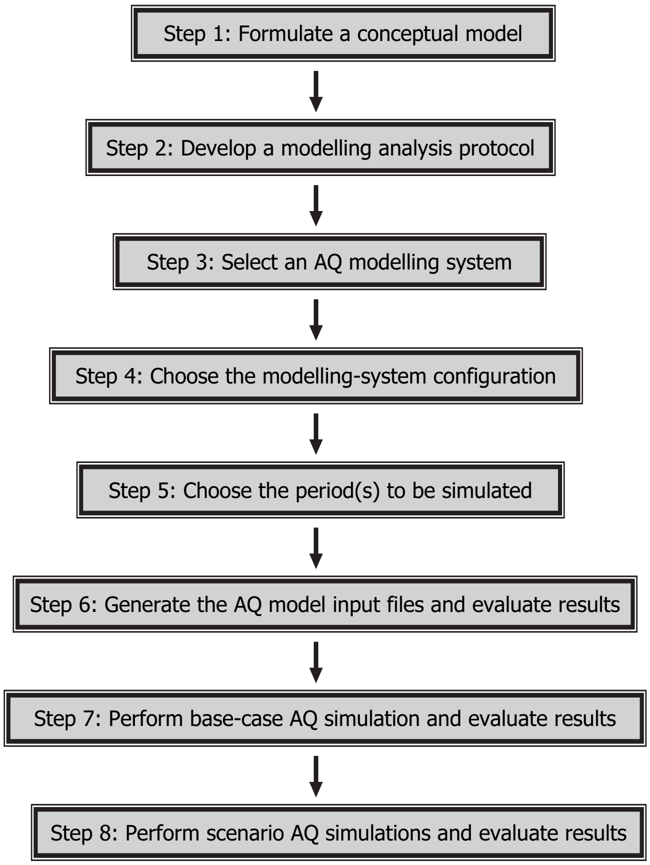



**FIG. 13.** Eight-step set of best practices for air quality modeling applications for emission control strategies.

considered with an AQ model. A conceptual model will provide useful guidance for all of the remaining modeling-process steps, including the identification of stakeholders, the selection and configuration of the AQ model, the development of candidate emission control scenarios, and the assessment of model results. As an example of how to formulate a conceptual model, Section 8 of U.S. EPA (2005c) lists a large number of questions about and analyses of emissions data, measurement data, and AQ model results that could be considered in developing a conceptual model for the occurrence of high annual or daily ozone levels in a particular locale or region.

2. *Develop a modeling/analysis protocol.*

   This step may be handled either formally or informally, but if model results are to be provided to interested parties, it is desirable early on to identify all interested parties and to obtain agreement on (a) which questions should be addressed, (b) what assumptions are reasonable to make (e.g., What processes can be neglected? How large does the model domain need to be?), (c) how the modeling work should be performed, (d) what sorts of results should be generated, (e) who should review them and how, (f) what the timetable should be, and so on. The conceptual model from Step 1 can guide this development, for example, by suggesting whether long-range transport is likely to be important or not, which will help to define the geographic extent of the "community" of stakeholders. Section 9.2 of U.S. EPA (2005c) gives a detailed suggested outline of what such a protocol might look like.

3. *Select an AQ modeling system.*

   Once the questions of interest have been identified in Step 2, the next step is to select an AQ modeling system that is capable of answering those questions and that has been judged to be credible and skillful through peer review and performance evaluations. Availability of model source code, previous successful applications to similar problems, and advanced science features and tools are useful additional selection criteria. Also, the time and resources available for the model application are practical considerations that may also affect the choice of modeling system. [McKeen et al. (2005) describe seven current PM AQ modeling systems. Other AQ modeling systems are described by EMEP (2005), Heald et al. (2005), Hodzic et al. (2006), Mallet and Sportisse (2005), Meng et al. (1997), and Zunckel et al. (2006).]

4. *Choose the modeling-system configuration.*

   Selection of an AQ modeling system is not sufficient by itself. It is also necessary to choose a modeling domain and map projection, horizontal and vertical grid resolution, level of nesting if any, an integration time step, a "spin-up" or "ramp-up" time, a "refresh" rate for the meteorological model, methods to specify chemical initial and boundary conditions, and, where choices are available, the particular physics and chemistry process parameterizations to be used in the meteorological and AQ models (e.g., convective parameterization, gas-phase chemistry mechanism, secondary organic aerosol scheme). In the case of an "off-line" AQ model, it is also desirable to harmonize to the extent possible the AQ model domain, map projection, and horizontal and vertical resolution with those of the companion meteorological model. Many of these choices will be guided by the conceptual model from Step 1 and the question(s) to be answered from Step 2. For example, the relative importance of long-range transport and the role, if any, of local terrain-forced meteorological circulations such as sea–land breezes will need to be considered. Sections 12 and 13.2 of U.S. EPA (2005c) provide some useful discussions about some of these choices.

5. *Choose the period(s) to be simulated.*

   This is one of the most open-ended steps, but it will be strongly constrained by the question(s) to be answered and, if relevant, by the form of the AQ standard (e.g., daily or annual, average or maximum) or the exact wording of the legislation of interest. In the case of short-term effects or AQ standards, the conceptual model should provide useful guidance, particularly related to the meteorological conditions that are associated with AQ exceedances. When choosing short-term simulation periods, Section 11 of U.S. EPA (2005c) recommends choosing a set of periods (a) for which extensive emissions, meteorological, and AQ data sets exist, (b) that correspond to a variety of relevant synoptic conditions,

and (c) that provide enough samples to have statistical significance, (d) where each period is long enough to span a full synoptic cycle (~5–15 days) and includes a relevant exceedance. By considering full synoptic cycles, the model is forced to simulate the conditions before and after an exceedance as well, allowing confirmation that the model can forecast nonexceedances as well as exceedances (e.g., Biswas et al., 2001). Additional reasons for choosing specific periods include (e) periods during intensive AQ field experiments, for which more detailed diagnostic evaluations can be performed, and (f) periods that have already been modeled, so that either model performance is already known to be satisfactory or else comparable results are available for comparison from a peer AQ model.

In the case of long-term effects or AQ standards, correspondingly longer simulation periods will be required. Continuing advances in computer technology have meant that running AQ models for periods as long as a year or more has become feasible (e.g., Eder & Yu, 2006), but even so there are still likely to be representativeness issues due to interannual meteorological variability (e.g., Brook & Johnson, 2000). Choosing periods that satisfy short-term selection criteria (a), (e), and (f) is desirable.

6. *Generate the AQ model input files and evaluate results.*

This step builds upon the previous four steps and will usually require (a) preparing geophysical fields for the selected domain and grid, (b) running a prognostic meteorological model with some type of data assimilation for the simulation periods selected in Step 5 to prepare meteorological input files, and (c) running an emissions processing system for the same simulation periods to prepare emissions input files for a base case and often a number of emission scenarios as well. For any regional (i.e., limited-area) AQ modeling system, it may also be necessary (d) to run both global meteorological models and AQ models or to analyze available chemical climatologies (e.g., Logan, 1999) in order to provide chemical boundary conditions. In preparing the input emission files, the size of the model domain will dictate how many emission inventories will need to be processed. For many North American model domains, it will be necessary to process both Canadian and U.S. or both U.S. and Mexican inventories, or in some cases, all three. For AQ modeling elsewhere in the world, such as East Asia or Europe, it is also likely that multiple national inventories will need to be combined.

As part of this step, it is also important to check the input files produced so as to ensure that the inputs provided to the AQ model are as accurate and credible as possible. As discussed before, meteorology drives the AQ simulation and the AQ model results are very sensitive to the meteorological inputs in complex and nonlinear ways. At a minimum, an operational evaluation should be performed against meteorological measurements: The suite of meteorological parameters considered should include temperature, humidity, wind speed, wind direction, cloud-related fields, precipitation, and, if possible, planetary boundary-layer depth (e.g., Hogrefe et al., 2001a; Smyth et al., 2006b). Evaluation of the processed emissions is not as straightforward, but current emissions processsing systems produce a range of log files and summary tables that can be checked for warning and error messages and for continuity, consistency, and plausibility, particularly when data from more than one country or jurisdiction are being combined. Visualization tools can also be applied to check the spatial and temporal patterns contained in the processed emission files. The emission files for various emissions scenarios should probably receive even greater scrutiny since extensive manipulations were likely required to transform current inventories to account for various socioeconomic projections and control measures. The inclusion or exclusion (depending upon the modeling/analysis protocol) and the treatment of natural emissions such as wildfires and windblown dust should also be checked. Sections 13 and 14 of U.S. EPA (2005c) provide useful and detailed discussions concerning this step.

7. *Perform base-case AQ simulation and evaluate results.*

The selected AQ model should have already undergone performance evaluations, but these may have been for other time periods. In this step the AQ model is run for the base case for the time periods selected in Step 5 and its performance is evaluated so as to characterize and quantify the overall modeling system's performance (i.e., including the treatment of emissions and meteorology) and to determine whether that performance is acceptable. Given known model limitations, errors, and uncertainties, Russell and Dennis (2000), Reid et al. (2003), Seigneur and Moran (2004), and the U.S. EPA (2001, 2005c) have all argued that this performance evaluation for the base case should not be restricted to just a basic operational evaluation against surface measurements of one or two pollutants, but instead should include a broader set of analyses that all feed into a "weight-of-evidence" judgment. Clearly, such an evaluation is somewhat open-ended and not prescriptive, but it should be more likely to lead to a correct judgment.

This broader set of analyses, many of them independent tests, could include any of the following possibilities:

- A more comprehensive operational evaluation, including consideration of a suite of ozone and PM precursors and other related gas-phase species (e.g., $NO_x$, $NO_y$, CO, $NH_3$, $H_2O_2$, $HNO_3$, individual VOC species) and PM chemical components both at the surface and aloft (e.g., Biswas et al., 2001).
- Sensitivity tests based on alternate configurations of the AQ model, including the use of a different emissions processing system or meteorological model, different rate constants and other model parameters, different grid resolutions, different chemistry mechanisms, and different boundary conditions (e.g., Mallet & Sportisse, 2006).

- Bounding tests in which emissions inputs are either increased or decreased to reflect the magnitude of uncertainties related to those inputs.
- Comparisons with results from peer AQ models, including operational AQ forecast models, if these have been run for the same region and time period(s) (e.g., Hogrefe et al., 2001b; Biswas et al., 2001; McKeen et al., 2005).
- If appropriate, comparison with receptor-based model results (e.g., Marmur et al., 2006).
- Comparison with observation-based models or analyses for chemical regimes, including indicator species ratios and gas ratio (e.g., Sillman et al., 1997; Stein & Lamb, 2002; Martin et al., 2004).
- Use of model probing techniques, including process analysis and direct decoupled method (e.g., Zhang et al., 2005).

8. *Perform scenario AQ simulations and evaluate results.*

   In this last step, once the scenario simulations have been performed and the results analyzed, several additional diagnostic or comparative evaluations can be carried out to examine the reasonableness of the AQ model's response to specified emission changes, particularly if disbenefits as well as benefits are predicted to occur. These include (a) applying observation-based models for chemical regime, model probing techniques, and sensitivity/bounding tests to the scenarios, (b) comparison with the relative response functions predicted by peer AQ models for the same set of scenarios, and (c) retrospective analyses of model response to historical emission changes.

   Seigneur and Moran (2004) described one comparative evaluation in which the predictions of two different PM AQ models were compared for the same three emission-change scenarios. Although the magnitudes of the responses for ozone, particulate nitrate, and $PM_{2.5}$ mass were all different between the two models for the three scenarios, the directions (i.e., sign) of the responses were the same, providing support for the general conclusions about atmospheric response. This directional consistency was particularly important in the scenario in which VOC emissions were reduced by half: Both models predicted a ~30% decrease in ozone levels (at one station) but an increase in both particulate nitrate and $PM_{2.5}$ mass levels, that is, a $PM_{2.5}$ disbenefit.

The effort required to follow best AQ modeling practice and to carry out each of the preceding eight steps for a model application may seem overwhelming. It is worth noting, however, that this is the worst case. For a jurisdiction with a history of AQ problems, a conceptual model (Step 1) likely already exists, and some AQ modeling may have already been performed. If an active in-house or external AQ modeling team with past experience for that jurisdiction can be accessed and the AQ modeling system that they use is credible, then Steps 2, 3, and 4 may not be needed and the modelers can begin at Step 5. If a new scenario is similar to a past scenario in terms of the periods to be simulated (Step 5) or most assumptions about emissions (Step 6), then the generation of input data sets likely will not require as much effort as a completely new scenario for a new period and/or new model domain. And if the base scenario has been considered before, then Step 7 may not be required either, so that the completion of Step 8 is effectively the minimum requirement for a new modeling study.

Furthermore, given the open-endedness of some of the above steps and the reality of limited resources, it may not be possible to do as thorough a job as policymakers and modelers would like to do. The penalty for "cutting corners" could but may not be incorrect predictions, but at a minimum it will be a greater degree of uncertainty and lower confidence in those predictions. Application of an AQ modeling system always entails many compromises, and the work that can be performed for the resources that are available is just one more compromise. However, the eight-step set of modeling best practices just described should be viewed as a goal to be approached as closely as possible if AQ modelers are to provide their clients with the best possible guidance.

## Combining Measurements, Emissions, and Model Output

Independently, emission inventories, measurement programs, and models are essential tools for AQ risk management and for describing the state of the atmosphere. A range of new methods is being explored, methods that combine emissions and measured and modeled concentration fields to expand the capability of daily, routine AQ forecasts and improve estimates of intra-urban and inter-urban variation in long-term or chronic exposure. "Fusing" these diverse information sources together to support a wide range of health and air quality studies, as well as real-time data reporting and analysis, holds considerable promise. Figure 14 presents a conceptual picture of the types of multiscale information that can potentially be assimilated or "fused" into a complete picture of the spatial variation in air pollutant concentrations. Although they are a source of input to the AQ models, emission inventories may also represent an independent source of spatial information and/or a predictor for use in empirical models.

Data assimilation routines using real-time data and model output are now being applied on a continuous basis to characterize large scale patterns across North America. The amount and quality of information available varies from pollutant to pollutant and geographically. At present, in North America, ozone is the most advanced, while routines for $PM_{2.5}$ are being developed. Figure 15 presents ozone concentrations across eastern North America derived from the Canadian Meteorological Centre air quality forecasting system and ozone data compiled under the AirNow program (Ménard & Robichaud, 2005). This image is derived from gridded ozone concentrations that are produced by combining observations with model-predicted concentrations to "interpolate" and prepare the data

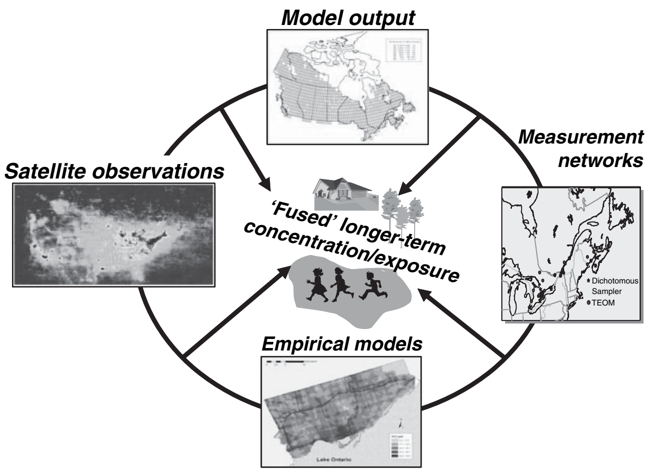



**FIG. 14.** Conceptual system of fused or assimilated data for estimating longer-term ambient concentration patterns or chronic air pollutant exposure at any geographic location and at any scale, from regional to neighbourhood. The system needs to be flexible in terms of which source(s) of information the estimate relies most heavily upon at each scale of interest and must be capable of providing estimates of location-dependent uncertainty. This uncertainty will vary geographically due to inconsistencies in the amount of information that is available at the finer scales.

for computing future concentrations using an air quality forecasting model. Hourly concentrations at each grid point were used to compute the maximum 8-h average concentration at each point on each day. These concentrations were then combined for the 5-mo period known as the "ozone season."

At present AQ models only assimilate surface observations, but approaches for "chemical data assimilation" are undergoing considerable research and development (Ménard, personal communication). The long-term goal is to begin utilizing observations from satellites and possibly other irregular sources of information (e.g., aircraft). The most advanced satellite instrument is OMI (Ozone Monitoring Instrument) on the Aura spacecraft, which was launched in 2004 (Schoeberl et al., 2004). In terms of the common air pollutants, daily, $13 \times 24$ km resolution observations for $O_3$, $NO_2$, $SO_2$ and aerosols are being measured. Devising the appropriate procedures for assimilating and/or interpreting such data presents a significant scientific challenge. Even with a satellite such as Aura, observations are available, at best, once per day if no clouds obscure the measurements.

The map of $NO_2$ over the northeast of North America shown in Figure 16 provides an indication of the capabilities of OMI. It is important to note that satellite observations of trace

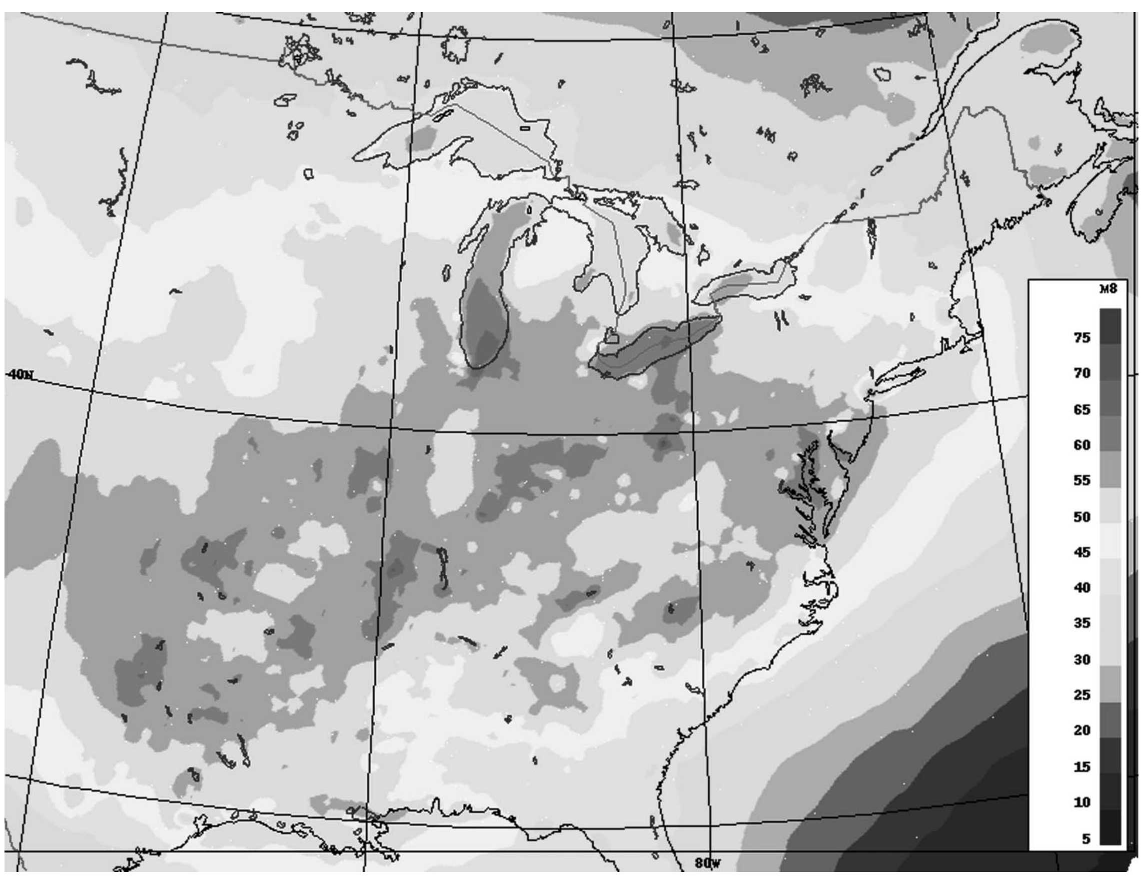



**FIG. 15.** Average daily 8 hour maximum ozone (ppb) for summer (May–Sept.) 2005.

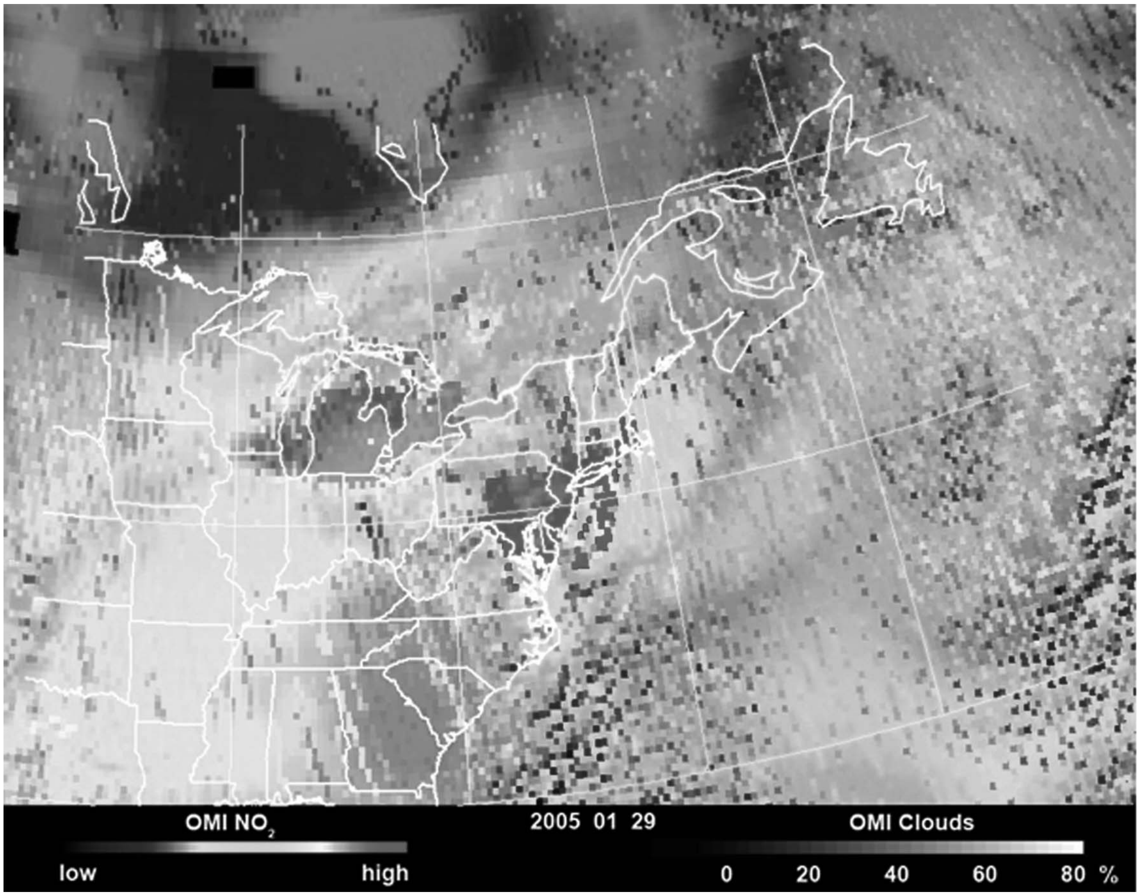



**FIG. 16.** Nitrogen dioxide ($NO_2$) observations from the ozone monitoring instrument (OMI) on the AURA satellite. The darker areas extending from Chicago to Toronto centered over Michigan and also on the US East Coast centered over New Jersey, show regions of high $NO_2$ concentrations on Jan. 29, 2005. On subsequent days through February 7, 2005, there was a large $PM_{2.5}$ event coinciding with some of these areas of high $NO_2$. This period represents the first wintertime air quality advisory ever issued in the Province of Ontario. Air quality alerts were also issued for Michigan during this period. The large concentrations over southern Michigan and southwestern Ontario are consistent with the surface conditions observed during the period. The OMI instrument was provided to the Aura Mission by the Dutch and Finnish Space and Meteoroligical Agencies. Image generated by OMI team (contact James F. Gleason). OMI $NO_2$ image provided by NASA. OMI is a joint effort of KNMI, NASA, and FMI, and is managed by NIVR and KNMI in the Nertherlands.



gases and aerosols are "total vertical column" amounts (i.e., not necessarily surface conditions). Research is needed to further improve the processing of satellite data from the raw signals and from other supporting data (e.g., correcting for clouds and variations in surface albedo) and then in deriving boundary-layer and/or surface concentrations, as well as vertical profiles. The latter of these requires, in itself, the combined use of AQ and meteorological models and surface observations.

Nonetheless, satellite data represents a valuable source of information because it is freely available and provides global coverage: Air pollutant information can be obtained where no monitoring exists. In addition to the initialization of AQ models, the spatial patterns derived from satellite observations (i.e., across days, weeks, or months) are well suited to determining, in an internally consistent manner, gradients in chronic exposure across large regions and among different countries. Thus far, aerosol observations ($PM_{2.5}$) have received the most attention for this purpose (e.g., Liu et al., 2005).

Population exposure to ambient air pollution occurs at neighborhood scales. This is beyond the resolution of all the sources of information discussed above (i.e., Figures 14, 15, and 16). Furthermore, it is unlikely that any of these will be able to resolve such scales in the future, and in terms of the types of deterministic AQ models discussed earlier, it is not reasonable to expect the meteorology to be modeled or for emissions information to be provided at a fine enough scale. At best, new parameterization schemes or independent emissions models will be developed to treat subgrid scale features to enable AQ models to reliably predict at the 1 to 5 km resolution. However, research is needed to determine how best to use such models to predict population exposure changes and their uncertainties within these grid sizes so that the costs and benefits of local-scale air quality risk management strategies can be evaluated.

To resolve urban to neighborhood scale exposure patterns for health studies, various approaches are currently being applied. These range from interpolation of monitoring site data

(e.g., Jerrett et al., 2005) to exposure surrogates such as distance-to-roadway and traffic counts (e.g., Hoek et al., 2002) to small-scale dispersion models and/or combinations of both (Wu et al., 2005; Cyrys et al., 2005). The local-scale exposures of interest have generally been associated with traffic, since data on road networks are readily available. However, a wider range of emission sources has been included in some ambient air pollutant exposure modeling efforts (e.g., Gram et al., 2003).

Geographic information systems (GIS) have proven to be useful for mapping exposure patterns, integrating different sources of information and in developing land-use regression models (Brauer, personal communication). Intra-urban chronic exposure estimates have been derived using land-use regression (LUR) for several cities (Brauer et al., 2003; Kanaroglou et al., 2005; Sahsuvaroglu et al., 2006; Gilbert et al., 2005; Silva et al., personal communication; Setton et al., personal communication; Luginaah et al., 2006; Brauer, personal communication). Cyrys et al. (2005) compared both LUR and dispersion model estimates for $NO_2$ and $PM_{2.5}$ and reported that for their cohort of interest in Munich, Germany, the two approaches led to similar exposure classifications. These results and most other LUR efforts have focused on traffic-related pollutants (e.g., $NO_2$ and sometimes $PM_{2.5}$). However, recent studies in Windsor, Ontario, have expanded the dependent variables in LUR to include $SO_2$, benzene, and toluene (Wheeler et al., 2006).

The empirical model image in Figure 14 is an example of the $NO_2$ surface predicted by LUR for Toronto, Ontario (Kanaroglou et al., 2005). The small scale spatial variability (i.e., neighbourhood scale or better) produced by applying the LUR for all points in a GIS database appears more realistic compared to the pattern obtained using interpolation, with respect to the known distribution of traffic (Jerrett et al., 2007b). A LUR model, once developed for the area of interest, also provides the capability of estimating chronic exposures for each member of a health study cohort if their addresses are known. Ideally, such estimates should be spot-checked with independent measurements within residential areas, inside a variety of homes and also in comparison to personal exposure measurements. This could potentially lead to the coupling of LUR models for outdoor, at-home concentrations with individual exposure models that consider home characteristics and time activity.

## Conclusions

Improving or maintaining air quality is a science- and technology-based activity, requiring governmental commitment to invest in the tools needed to reach informed decisions. In jurisdictions where AQ management has not been a priority, solutions to the current problems may be relatively straightforward, such as eliminating local/residential burning for cooking and heating fuels. However, even when all the obvious and/or cost-effective measures have been implemented, AQ problems can still persist and even become worse due to economic and/or population growth. In this situation, which may be the case for many developed countries, the best approaches to improve AQ are not as easy to identify. Accurate, comprehensive emission inventories, AQ measurements, and models are therefore essential to make headway. However, they need to be applied intelligently, following, as much as possible, best practices as informed by experience. This includes a well-developed conceptual model of the relationship between emissions and observations. To make headway, air quality targets (or standards or objectives) are also needed and clearly, public health protection is one of the main motivations behind such targets. This necessitates establishing a quantitative link between air pollutant levels and health impacts, such as a concentration-response relationship, which is dependent upon the availability of measurements. However, there is a need to develop air quality risk management methods that integrate multiple environmental issues, not just human health concerns.

This section reviewed several key issues related to the development, use and improvement of emissions, measurements, and models for AQ management. Each can provide useful information for AQ risk management, and when they are considered together they can provide additional insights and guidance. However, AQ models depend upon the availability of information about measurements, emissions, and meteorology, whereas the converse is not true. AQ modeling should thus follow and not precede the development of measurement and emissions information.

Accurate emission inventories are the foundation of all air quality management programs. They provide the essential information needed to understand the effects of air pollutants on human and ecosystem health, to identify which sources need to be controlled in order to protect health and the environment, and they provide the information needed to determine whether or not actions taken to reduce emissions have been effective.

In principle, the development of emission inventories would seem to be a relatively straightforward process, but in practice their production is found to be a very complex and demanding task. As initial actions to reduce emissions from large point sources find success, understanding and addressing residual air quality problems requires greater effort and emission inventories of increasing sophistication. Fortunately, much has been learned over the past 40 yr that can make the development of new inventories a more systematic process. New measurement technologies and better understanding of the chemistry and physics of pollutant formation will continue to multiply the number of sources that can be measured directly and assure that these measurements reflect what is actually entering the atmosphere. Likewise, new methods for deducing and characterizing uncertainty will result in better understanding of the accuracy with which we know primary and precursor emissions. Finally, better data management software, the ubiquitous availability of low-cost, high-end computing, and growing

availability of high-bandwidth communications have made the development, maintenance, dissemination, and use of large data sets practical for nearly everyone.

Air quality measurements are essential for public health protection. They define the problem to be managed, and serve as the basis for determining the current level of health risk a given population is experiencing and consequently for prioritizing the need for reductions. Measurements are also critical for evaluating the effectiveness of AQ management strategies and altering such strategies if the desired outcomes are not being achieved.

Detailed analysis of measurement data can help target the most effective approaches to reduce ambient concentrations and, hopefully in the future, the optimum (i.e., cost-effective) approach to protect public health. Full understanding of particulate matter, in terms of impacts, formation processes, and optimal control strategies, remains the air pollutant area requiring the greatest attention in terms of detailed measurement studies and data interpretation efforts.

When measurement programs are forward looking, pushing the limits of what can be routinely monitored, they can provide new insights regarding additional air pollutants of concern and can support future epidemiological studies to uncover new risks to the population. Ultimately, the availability of air quality measurements dictates what can be studied, and thus there is a continual need to expand the pollutants measured and the location and temporal resolution of such measurements. Combining the data from a variety of measurement approaches, including remote sensing, with the data from both physical and empirical models provides an improved picture of spatial and temporal patterns. These improvements are providing better AQ information to scientists, the public, and decision makers and ultimately can be expected to lead to a better understanding of AQ impacts and of possible approaches to protect public health.

AQ models are able to quantify the links between emissions of primary pollutants or precursors of secondary pollutants and ambient pollutant concentrations and other physiologically, environmentally, and optically important properties. They are the only tool available that can predict, based on possible future emission levels, future spatially and temporally resolved air concentration and deposition patterns and that can address multiple pollutants simultaneously and quantify possible co-benefits. AQ models can also account for the impacts of nonlinear processes and are able to predict whether a candidate abatement strategy will lead to benefits or disbenefits or both.

There are a large number of possible sources of AQ model error and uncertainty, ranging from not understanding the underlying science and truncation errors intrinsic to the numerical techniques employed by the model, to uncertainties in the input emissions and model–measurement incommensurability. There are even more ways for these numerous sources of error and uncertainty to interact, often nonlinearly and sometimes canceling out (so-called “compensating errors”). As a consequence, AQ model uncertainty is impossible to quantify definitively, but it is possible to characterize it through model performance evaluations, model intercomparisons, and sensitivity and bounding tests. There are also varying degrees of uncertainties across pollutants and their components.

Box (1979) wrote that “All models are wrong, but some are useful.” For an AQ model to be “useful,” it should be credible. For it to be credible, it should give the right answers for the right reasons. AQ model credibility is established through model review, model performance evaluations, and successful model applications. However, model credibility is always provisional, so model evaluation (and model improvement) should be an ongoing process.

A key question in applying AQ models is: “How accurate does a model need to be?” Reid et al. (2003) suggested that the general answer is that “model predictions should be good enough that model uncertainty does not affect the decisions that are based on the predictions.” AQ modeling uncertainty can be managed and limited by following “best practice” at all stages of the modeling process. Best practice basically boils down to thoughtful and careful selection, set up, and application of a credible AQ modeling system accompanied by careful scrutiny and consistency checking of the results by various means, including measurements and results from both alternate configurations of the selected AQ model and from other AQ models. The credibility of the model predictions for a given application is then determined based on a weight-of-evidence judgment that considers all of the evaluation results. This process is not at all “cut and dried”—it is much more in the nature of applied research than a routine activity.

### Key Messages

- Three essential tools for managing the risk due to air pollution are multipollutant emission inventories, ambient measurements, and air quality models. Tremendous advances have and continue to be made in each of these areas, as well as in the analysis, interpretation, and integration of the information they provide.
- Accurate emission inventories provide essential information to understand the effects of air pollutants on human and ecosystem health, to identify which sources need to be controlled in order to protect health and the environment, and to determine whether or not actions taken to reduce emissions have been effective.
- Air quality measurements are essential for public health protection and are the basis for determining the current level of population health risk and for prioritizing the need for reductions. They are also critical for evaluating the effectiveness of AQ management strategies and altering such strategies if the desired outcomes are not being achieved.
- Air quality models quantify the links between emissions of primary pollutants or precursors of secondary

pollutants and ambient pollutant concentrations and other physiologically, environmentally, and optically important properties. They are the only tool available for detailed predictions of *future* air concentration and deposition patterns based on possible future emission levels and climate conditions.

- Air quality problems tend to become more difficult to address as the more obvious and less costly emission control strategies are implemented. This increases the demand for advanced scientific and technological tools that provide a more accurate understanding of the linkages between emission sources and ambient air quality.
- Despite scientific advancements, including improved understanding of the impacts of poor air quality, the pressure to identify cost-effective policies that provide the maximum benefit to public health pushes our current tools and knowledge to their limits and beyond.
- Due to scientific uncertainties, highly specific control options that target specific chemical compounds found on fine particles, specific sources, or source sectors or that lead to subtle changes in the overall mix of chemicals in the air (gases and particles) remain extremely difficult to evaluate in terms of which options most benefit public health. Lack of a complete understanding of exposure and health impacts of the individual components in the mix and their additive or synergistic effects poses further challenges for health benefits evaluation. However, progress is being made and new ways of thinking about air quality and pollution sources, such as the concept of intake fraction, help to provide some perspective.
- A broader perspective, including consideration of environmental effects and the implications of climate change on air quality and on co-management of air pollutants and greenhouse gases, will be increasingly important to embrace.

## AIR QUALITY MANAGEMENT APPROACHES AND EVIDENCE OF EFFECTIVENESS

### Introduction

This section focuses on describing how air pollution problems are managed in North America, within the European Community (EC; former European Economic Community, EEC), and in Asia. Policy approaches are reviewed, including mobile source, point source, and area source emission reduction strategies; standard-setting approaches; market-based approaches; trans-boundary strategies; multipollutant strategies; and public education/behavioral approaches. Case studies of air quality management in large urban centres within each continent provide more detailed examples to illustrate the mix of strategies and their impact on air quality. The section concludes with evidence from intervention studies to illustrate the public health benefits associated with reductions in pollutant emissions.

### Air Quality Management in North America

This section provides a perspective on air quality management in North America, focusing on the past and present situation in the United States, Canada, and Mexico. For each country, the historical development of clean air policies and programs is provided, as well as a brief description of major emissions sources, an overview of some of the main regulatory and nonregulatory air quality management initiatives, and trends in ambient air concentrations as an indicator of overall program effectiveness.

Within North America, each country sets separate ambient air quality standards (see Table 4). Within the United States, California has set its own standards, generally more

**TABLE 4**
Ambient air quality standards for North America

| Pollutant | Averaging period | U.S. | California | Mexico | Canada[a] |
|---|---|---|---|---|---|
| $SO_2$ | 1 hour | – | 655 μg/m$^3$ | – | 172 μg/m$^3$ |
| | 1 day | 365 μg/m$^3$ | 105 μg/m$^3$ | 340 μg/m$^3$ | 300 μg/m$^3$ |
| | 1 year | 80 μg/m$^3$ | – | 80 μg/m$^3$ | 60 μg/m$^3$ |
| $NO_2$ | 1 hour | – | 338 μg/m$^3$ | 400 μg/m$^3$ | 400 μg/m$^3$ |
| | 1 year | 100 μg/m$^3$ | 56 μg/m$^3$ | 400 μg/m$^3$ | 100 μg/m$^3$ |
| $PM_{10}$ | 1 day | 150 μg/m$^3$ | 50 μg/m$^3$ | 150 μg/m$^3$ | 50 μg/m$^3$ |
| | 1 year | – | 20 μg/m$^3$ | 50 μg/m$^3$ | – |
| $PM_{2.5}$ | 1 day | 35 μg/m$^3$ | – | – | 30 μg/m$^3$ |
| | 1 year | 15 μg/m$^3$ | 12 μg/m$^3$ | – | – |
| Ozone | 1 hour | – | 180 μg/m$^3$ | 216 μg/m$^3$ | 160 μg/m$^3$ |
| | 8 hour | 150 μg/m$^3$ | 140 μg/m$^3$ | – | 130 μg/m$^3$ |
| CO | 1 hour | 40 mg/m$^3$ | 23 mg/m$^3$ | – | 36 mg/m$^3$ |
| | 8 hour | 10 mg/m$^3$ | 10 mg/m$^3$ | 13 mg/m$^3$ | 15 mg/m$^3$ |

[a]Canadian values are based on Maximum Acceptable Levels.

stringent than those set by the U.S. Environmental Protection Agency.

Even though they rely on many of the same human exposure and epidemiological studies, these standards have striking differences. The treatment of allowances, spatial averaging over monitors and special events, such as dust storms and prescribed burns can greatly affect the stringency of standards.

*Air Quality Management in the United States*

This description is based largely on Cote et al. (2008) and National Research Council (2004).

*Historical Perspective on Air Quality Management in the United States*

Air quality control was first addressed by the U.S. federal government in the 1955 Air Pollution Control Act and its 1959 extension, which provided money to state and local agencies for research and training on air quality. This was followed by a series of acts including the 1963 Clean Air Act. However, it was in 1970 that two fundamental events set the stage for subsequent air quality management (AQM) approaches in the United States: the creation of the U.S. Environmental Protection Agency (EPA) by President Nixon, and significant amendments to the Clean Air Act (CAA).

The introduction to the CAA lists four overarching goals:

- To protect and enhance the quality of the nation's air resources so as to promote the public health and welfare and the productive capacity of its population.
- To initiate and accelerate a national research and development program to achieve the prevention and control of air pollution.
- To provide technical and financial assistance to State and local governments in connection with the development and execution of their air pollution prevention and control programs.
- To encourage and assist the development and operation of regional air pollution prevention and control programs.

The 1970 CAA authorized the U.S. EPA to set national ambient air quality standards (NAAQS) for criteria air pollutants, defined as those "in the ambient air resulting from numerous or diverse mobile or stationary sources." It also allowed for emissions standards for hazardous air pollutants, development of aircraft emissions standards, some automotive emissions standards, and motor vehicle emissions inspection and maintenance programs. The 1970 CAA also allowed individual states to take over responsibility for compliance with the CAA in return for funding. In order to receive the funding, states submitted state implementation plans (SIPs) describing plans to meet the U.S. EPA requirements. SIPs affect mainly local areas where pollution levels exceed the standards, and usually include control of large industrial sources. States were also granted permission to adopt air quality guidelines that were more stringent than federal standards. Once a SIP receives approval from state and federal regulatory bodies it becomes legally enforceable at both levels.

In 1971, initial NAAQS were established for CO, $NO_2$, $SO_2$, TSP, hydrocarbons, and photochemical oxidants. The U.S. EPA was tasked with reviewing each of the NAAQS every 5 yr. At the same time, a U.S. ambient air quality monitoring program began. Over time the agents defined as criteria air contaminants have changed somewhat: Lead was added in 1976, and in 1979, "photochemical oxidants" was replaced by ozone. Hydrocarbons were removed in 1983, and separate standards for $PM_{10}$ and $PM_{2.5}$ have now replaced TSP.

The first prospective cohort studies to examine the relationship between air pollution and health, the Harvard Six Cities (Dockery et al., 1993) and the American Cancer Society (Pope et al., 1995) studies, were initiated in the early 1970s. The publication of their results in the 1990s provided important evidence that there was a significant association between living in a polluted city and risk of premature death. In response, the American Lung Association sued the U.S. EPA, declaring that the agency had failed to meet its obligation to review the NAAQS every 5 yr, and that the new evidence obligated the U.S. EPA to conduct a new review. The result was a court order resulting in an accelerated but contentious review of the PM standards. Ultimately, the U.S. EPA committed to a nationwide $PM_{2.5}$ monitoring network, Congress funded a new multifaceted research program, and the administration agreed not to implement the new standard until the next 5-yr review was completed in 2002.

Efforts to meet the NAAQS have not always resulted in attainment, but they appear to have contributed substantially to reductions in pollutant emissions across the United States. Limitations on continued improvement to achieve the NAAQS are imposed by growth in population, energy use, the number of sources, and vehicle miles traveled.

The initial regulations for hazardous air pollutants (HAPs) under the 1970 CAA authorizations were mainly national standards that were applied to specific industries. Between 1970 and 1990, only the eight hazardous air pollutants with the strongest evidence for harm were regulated (asbestos, benzene, beryllium, coke oven emissions, inorganic arsenic, mercury, radionuclides, and vinyl chloride). Efforts to expand the regulations to other substances were hampered by legal and scientific arguments over risk assessment methods and assumptions, the amount of evidence required to justify regulations, the cost to industry, benefits to human health and the natural environment, and debates over "how safe is safe" (National Research Council, 2004).

This lack of progress on regulation at the federal level led agencies at all levels to turn to the individual states for implementation of AQM. During the 1980s, federal grants, training, and technology transfer facilitated expansion of regional programs, and almost all regulation of HAPs occurred at the state

level. Although many of these programs continue to be strong, there is significant interstate variability in AQM approaches, likely due to the lack of national leadership during the 1980s.

In 1990, a new set of amendments to the CAA was adopted, largely in response to Congress's dissatisfaction with the lack of productive AQM for HAPs on a national scale. The amendments replaced the risk-based approach to managing the industrial sector with a technology-based approach. The amendments identified 189 HAPs for management, and defined sources where emissions standards should apply. The current list is intended to be periodically reviewed and amended as dictated by new scientific information. The amendments also addressed nonattainment areas, mobile sources, acid rain, permits, stratospheric ozone, and enforcement.

Beginning in the 1970s, acid rain, which results from the chemical conversion of $SO_2$ and $NO_x$ to sulfuric and nitric acid in the atmosphere, became a national concern. $SO_2$ is emitted primarily by coal-fired power plants, and $NO_x$ emissions are mainly a result of coal combustion in power plants and fuel combustion in vehicles. The site of acid deposition is typically distant from the point of emission because of the time it takes for atmospheric conversion of the gases to acid.

Pressure from states experiencing acid deposition and from the Canadian government led to funds for research into the impacts of acid rain and recommendations on whether emissions control approaches were required to mitigate them. The 1990 CAA amendments addressed acid rain in a form of legislation which represented a significant departure from previous approaches to regulating criteria air contaminants and HAPs: "cap and trade." This program (described in more detail later) set a maximum emissions level and assigned emissions allowances to individual emitters. Emitters were then allowed to design their own compliance strategies, which could involve trade in emissions allowances.

In 2004, the National Academy of Science identified seven challenges facing U.S. air quality management for the future:

- Achievement of standards—further reductions in emissions will be required in order to meet the 1998 standards for ozone and particulate matter as well as the 1999 regulations for regional haze.
- Toxic air pollutants—further research is needed on the sources, atmospheric transport and distributions, and health effects of toxics.
- Health effects at low pollution concentrations—there is increasing evidence that there is no level below which exposure to some pollutants has no potential health effects. This may have implications for how some pollutants are regulated.
- Environmental justice—there are currently no programs under the CAA that address mitigating pollution that might be disproportionately born by minority and/or low-income groups in densely populated urban areas.
- Protecting ecosystem health—protection of ecosystems affected by air pollution has received insufficient attention despite being mandated in the CAA.
- Multistate, cross-border, and intercontinental transport—air quality in a particular area can be affected by pollutant transport across geographic areas including political boundaries.
- AQM and climate change—AQM systems must ensure that pollution reduction strategies remain effective as the climate changes. Multipollutant approaches that include reducing emissions that contribute to both climate warming and air pollution may be desirable.

### *Regulation of Air Pollutants in the United States*

*The Clean Air Act.* In the United States, air quality management is undertaken by local, tribal, state, and federal authorities, with responsibilities delegated to each jurisdiction by the CAA. The U.S. EPA coordinates the federal government's role, which is to ensure a basic level of environmental protection across the country through national uniformity in air quality standards and pollution mitigation approaches. The CAA also charges the U.S. EPA with overseeing actions carried out by agencies at all levels, which may include imposing federal sanctions or federally developed pollution-control plans on delinquent areas. However, most of the responsibility for implementing federal rules and developing strategies and control measures to meet national air quality standards falls on state and local governments. An overview of air quality management activities is provided by Figure 17.

Federal rules promulgated under the CAA are subject to judicial review by the courts. Courts may set aside an agency rule if they find that it was not based on consideration of relevant factors, or if an error in judgment was made. However, the court may not substitute its judgment for the U.S. EPA's.

Five major goals are identified in the most recently amended CAA:

- Mitigating potentially harmful human and ecosystem exposure to six criteria air pollutants (CO, $NO_2$, $SO_2$, $O_3$, PM, and lead).
- Limiting sources and risks from exposure to hazardous air pollutants (HAPs, also called air toxics).
- Protecting and improving visibility impairment in wilderness areas and national parks.
- Reducing the emissions of species that cause acid rain, specifically $SO_2$ and $NO_x$.
- Curbing the use of chemicals that have the potential to deplete the stratospheric $O_3$ layer.

*Regulation of criteria air contaminants and hazardous air pollutants.* There are fundamental differences in the way that criteria air contaminants and HAPs are regulated. The six pollutants regulated under NAAQS (the criteria air contaminants) are considered to originate from multiple sources and are characterized as being more ubiquitous and therefore having a

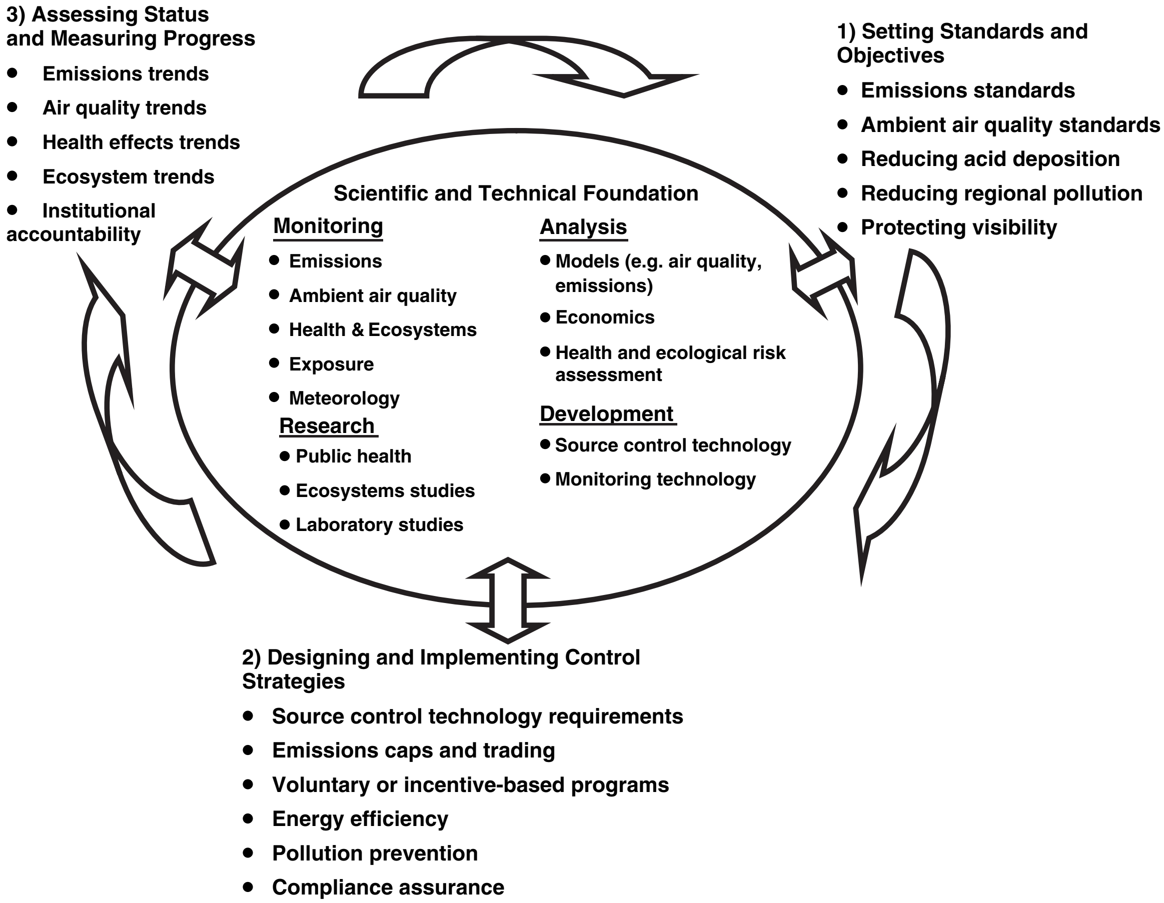



**FIG. 17.** Schematic of U.S. Air Quality Management Activities. (Cote et al., 2008). Reprinted by permission of the publisher (Taylor & Francis Ltd.).



greater impact on human health. These are regulated through ambient air concentration and time standards that define maximum allowable ambient concentration as well as monitoring and statistical methods to be used when determining if an area is in compliance. Two types of standards exist: primary standards, which are intended to protect the health of the most sensitive population subgroups with an adequate margin of safety, and secondary standards, which are intended to protect public welfare (by addressing issues such as visibility and ecosystem impacts). The CAA specifies the date by which primary standards must be met and gives the U.S. EPA authority to enforce compliance. Although reviews of the air quality data and NAAQS are ideally conducted every 5 yr, the complexity of the review process combined with unusually rapid advancement in evidence related to the stringency of the standards has resulted in more extended periods between reviews. The Supreme Court has determined that economic consequences should not be considered when setting a primary NAAQS, although costs are assessed during the NAAQS-setting process to meet the requirements of the President's Executive orders on the need for Cost-Benefit Analyses for new regulations and are also assessed for rules implementing the standards.

The process for attaining NAAQS includes monitoring ambient concentrations, designation of nonattainment plans, and implementation of SIPs. Although most pollution reduction activities therefore take place on a local or regional scale, uniform national regulations have been adopted for some stationary and mobile criteria air contaminant sources in an effort to avoid placing an unfair economic burden on individual states.

In contrast to criteria air contaminants, HAPs are regulated at the point of emission for stationary sources and area sources. Standards for HAPs are developed based on threshold for health effects, accounting for a margin of safety. Facilities emitting large quantities of HAPs, i.e., 10 tonnes or more of any individual HAP or a combined total of 25 tonnes or more on an annual basis, are defined as "major emitters." One hundred and seventy-four types of sources fall under this definition, and these facilities are required to implement maximum achievable control technologies (MACT) and work-based practices. Standards are also used to control area sources of HAPs, where standards must be imposed on a sufficient number of area sources so as to ensure that sources representing 90% of the area source emissions (excluding mobile sources)

of the 30 (or more) HAPs that pose the greatest threat in the largest number of urban areas are subject to regulation. For these sources, the U.S. EPA administrator chooses whether MACT or GACT (generally available control technologies) are more appropriate.

In cases of major stationary or area sources, regulation is followed by assessment of residual risk. After this assessment, if no action was taken by Congress for 2 yr, the U.S. EPA administrator should promulgate emissions standards to "provide an ample margin of safety to protect public health or to prevent, taking into consideration costs, energy, safety, and other relevant factors, an adverse environmental effect" (National Research Council, 2004). However, because of the difficulties in assessing residual risk, this process has not yet been completed.

SIPs are state implementation plans, which must be devised by each state under the CAA. These are dynamic plans that must evolve to meet new federal or state requirements, changes in status of NAAQS attainment, or address other new information. They must be submitted within 3 yr of a new NAAQS being promulgated, and provide "implementation, maintenance, and enforcement" of the standard. The specific requirements for each SIP depend on the state's air quality, which is determined by its compliance with NAAQS for criteria air pollutants:

- Nonattainment—any area (such as an urban center) that does not meet a primary or secondary NAAQS. $O_3$ nonattainment areas are further classified as being marginal, moderate, serious, severe, or extreme.
- Attainment—any area that meets the primary and secondary NAAQS and does not contribute to the violation of a primary or secondary NAAQS in a nearby area.
- Unclassifiable—any area that cannot be classified on the basis of available information.

Specific procedures are designated for determining attainment status and for setting requirements for nonattainment areas. Actions to be carried out in nonattainment areas include providing a plan for implementing reasonably available control technologies (RACT), meeting primary NAAQS, offsetting emissions from any new or modified major sources, and for installing MACT, comprehensive emissions inventories, and implementing new-source reviews before construction.

Data from national air quality monitoring networks suggests that SIPs have contributed toward air quality improvement. Drawbacks of the process include its overly bureaucratic nature, an overemphasis on modeled attainment demonstrations, and the single-pollutant focus. Another weakness of the SIP AQM approach is its inability to deal with pollutants crossing jurisdictional boundaries. One method that has been used to approach the problem is "cap and trade," an approach that was first applied to reducing acid deposition.

*Cap and trade.* Initially, the most cost-effective way to reduce the local impact of $SO_2$ and $NO_x$ emissions from stationary sources was to install taller stacks: an approach that significantly reduced local concentrations of the pollutants and helped urban areas attain the NAAQS. However, it resulted in long-range dispersion of the pollutants, contributing to the acid rain problem. Cap and trade is a market-based strategy for emissions reduction whereby an aggregate emissions cap for a particular pollutant is set by an agency such as the U.S. EPA, but discrete amounts of the pollutant are traded among sources.

In 1990, Congress implemented a cap and trade program on $SO_2$ in response to the acid deposition issue, allocating an annual emissions allowance to each electricity-generating facility. The allowance is determined based on historical resource consumption and entitles the holder to emit a defined amount of $SO_2$ each year. If the facility's emissions exceed the allocated emissions amount, the facility can either reduce its emissions to achieve the allowance, or purchase allowances from other facilities which have a surplus. The facilities must report their emissions regularly, and the U.S. EPA manages an allowance tracking system and ensures compliance.

The overall cap set by Congress represented a 50% reduction in emissions nationwide, which was believed to be sufficient to ensure that trading would not create regions of unacceptably high emissions. The $SO_2$ cap-and-trade program has resulted in rapid emissions reductions at lower cost than was initially anticipated. Features believed to be associated with its success include its simplicity, the availability of CEM (source continuous emissions monitor) systems, transparency, certainty of penalties, and the opportunity for banking emissions allowances.

In 2005, the Clean Air Interstate Rule (CAIR) was promulgated. It caps $SO_2$ and $NO_x$ emissions in 28 Eastern States and the District of Columbia, mandating the largest pollution reductions since those set by the acid rain program. States can achieve the emissions reductions by either (a) requiring power plants to participate in an interstate cap-and-trade system that will be administered by the U.S. EPA and cap emissions in two stages, or (b) meeting an individual state air emission limit through some measure chosen by the state. In 2005, the United States also promulgated the Clean Air Mercury Rule, which will permanently cap and reduce mercury emissions from coal-fired power plants.

*Management of mobile source emissions.* Several approaches exist to reducing emissions from mobile sources. One is new-source certification programs, which specify emissions standards that apply to new vehicles and motors. The 1970 CAA amendments required vehicle manufacturers to reduce light-duty vehicle (LDV) and light-duty truck (LDT) emissions by 90%. Although the CAA amendments in 1977 extended the deadlines after the manufacturing industry claimed that the time scale for implementation was too short, the approach reflected an important new "technology-promoting" approach by Congress. Over time, the development of the technology was refined and installed on new vehicles, and by the end of the

1980s, emissions control devices were widespread throughout the U.S. automotive fleet. Despite the success of the technologies, the increase in vehicle miles traveled and the discovery that some evaporative emissions were not being controlled continued to contribute to nonattainment for ozone across the United States. In 1990, amendments to the CAA mandated emissions reductions referred to as Tier 1 controls for LDVs. These called for further reductions in $NO_x$ and VOCs and tightening controls on evaporative emissions, including during refueling. Since then, Tier 2 standards have been promulgated, which tighten $NO_x$ and VOC emissions standards even further, limit sulfur content in fuel, and place regulations on medium-duty passenger vehicles.

A series of negotiations between concerned states, car manufacturers, environmental groups, and the U.S. EPA has resulted in a voluntary national low-emission vehicle (NLEV) program. Although the U.S. EPA sets regulations for the program, they come into effect only if and when states and auto manufacturers opt into it.

Heavy-duty vehicles (HDV) were regulated beginning in the 1980s, and in 2001 the United States adopted new regulations that require reductions in fuel sulfur content and tightening of emission certification standards. Nonroad engines, which encompass a wide variety of engines, including land-based diesel engines, spark ignition engines, marine engines, and diesel locomotive engines, have generally not been subject to emissions regulations in the United States. However, in 2004, the U.S. EPA finalized a national program to reduce emission from nonroad diesel engines through a combination of fuel and engine controls. The proposed standards would take effect as early as 2008 and are expected to reduce emissions by more than 90%.

Regulations on mobile sources also include in-use technological measures and controls, which includes specifications on fuel properties, vehicle inspection and maintenance programs, and retrofits to existing vehicles. Inspection and maintenance programs for LDVs and LDTs were enhanced after the 1990 CAA amendments but remain a controversial issue politically. This is partly a result of the U.S. EPA's use of a particular model (MOBILE) for estimating emission-reduction benefits from inspection and maintenance (I/M) programs. Although most states have implemented some form of I/M, there were many that initially did not meet all of the U.S. EPA's requirements. Thus, in 1995, Congress responded by allowing more flexibility. Technical controversies also hamper I/M programs, with recent allegations that the programs are not as effective at identifying faulty or noncompliant emissions in vehicles as originally thought.

Regulating in-use emissions of HDVs has also been difficult. The vehicles are typically sturdier and remain in use longer, meaning that older, more inefficient engines remain in operation much longer than for LDVs. Also, it is technically challenging to conduct accurate in-use emissions testing.

Beginning in the late 1980s, a strategy that combined vehicle performance with fuel quality was adopted. The phase-out of lead was highly successful, and the introduction of reformulated gasoline resulted in important reductions in benzene emissions. The federal reformulated gasoline program included performance requirements and content reductions. Implementation of sulfur-reduction regulations is ongoing.

Behavioral and societal strategies also constitute a method of reducing mobile source emissions. Although the 1970 CAA required states to develop transportation control plans (TCPs) for their metropolitan areas, the policies that would be required to attain the NAAQS by 1975 were severe and highly unpopular. Many states refused to submit TCPs, and over time, regulations on motor vehicle use in the states have continued to be politically unfeasible. Efforts to link air quality legislation to transportation planning and investment has met similar institutional resistance and difficulties. In 1990, the CAA amendments required tighter integration of clean air and transportation planning. This affected mainly metropolitan planning organizations (MPOs), the agencies that conduct transportation planning under federal law. If conformity is not maintained, federal funding for transportation can be cut off. If forecasted emissions result in an exceedance of permissible levels as defined by the SIP, the MPO must either alter its transportation plan or promulgate additional mobile or stationary source controls. This has had the greatest impact on rapidly growing urban areas where there is economic and political pressure to expand the transportation infrastructure.

*Case Study: Air Quality Management in California*

This description is adapted from O'Connor and Cross (2006).

*The role of the California Air Resources Board.* California's air pollution control program began in 1959, when the California legislature created the California Motor Vehicle Pollution Control Board, to certify emission control devices for vehicles. Subsequently, under the Federal Air Quality Act of 1967, California was granted a waiver to adopt and enforce its own emission standards for new vehicles, in recognition of its unique air quality and need to set more stringent emission control requirements compared to the rest of the nation. In 1967, the California Air Resources Board (CARB) was formed through the Mulford-Carrel Air Resources Act, signed into law by Governor Ronald Reagen. The Act created CARB by merging the California Motor Vehicle Pollution Control Board and the Bureau of Air Sanitation. CARB has the ability to set mobile source emission standards more stringently than the U.S. Environmental Protection Agency, except sources involved in interstate commerce: trains, planes, ships, and interstate trucking. Other states, like many in the northeastern United States, have taken advantage of their option to adopt California's mobile source emission standards.

CARB also sets regulations for consumer products, paints, and solvents, and identifies and controls toxic air contaminants.

It coordinates the efforts of federal, state, and local authorities to meet ambient air quality standards, while minimizing the impacts on the economy. While local air quality management districts have the primary authority to control emissions from stationary and areas sources, CARB can assume this authority if local agencies do not develop or implement their air quality plans. Californians support and want air pollution control—65% support environmental protection over economic growth (although California has accomplished both), and this has created a supportive legislature. For example, the California Legislature recently passed a bill (signed by Governor Arnold Schwarzenegger) to give CARB the authority to regulate greenhouse gas emissions 1990 levels by 2020, a 29% reduction from business as usual.

The governor of California, with the consent of the state senate, appoints the 11 members of CARB, 5 of which are from local air quality management districts. It is an independent board when making regulatory decisions. The board is required to have a medical doctor and an engineer as members. The first chairman was a respected atmospheric scientist (Professor Arie Haagen-Smit) who discovered how urban smog was created, and last year's (Dr. Robert Sawyer) was formerly a mechanical engineering professor at the University of California, Berkeley. Except for the chairman, the board only works once per month and relies on its staff for technical input. The board oversees a $150 million budget and a staff of over 1100 employees located in northern and southern California. In addition, the board provides financial and technical support to the 35 local districts. CARB is funded by vehicle registration fees and fees on stationary sources and consumer products. It also receives up to $166 million per year in incentive funds from fees on vehicle registration and new tire sales. This goes to diesel engine retrofits, car scrappage, and agricultural, port, and locomotive projects.

California has 4000 air quality professionals at the state and local levels. Most of CARB's workforce are engineers and scientists, and about 20% have PhDs and master's degrees. CARB conducts its own vehicle testing programs and funds extramural research at a level of $5 million per year, taking advantage of the strong academic community in California and other states. It also funds a technology demonstration and commercialization program, and the development of state-of-the-art emission, air quality, and macroeconomic models. The technology research demonstrates that reduced emissions are feasible, but the use of performance-based standards allows industry to come up with more cost-effective approaches. Enforcement and monitoring programs ensure that the emission standards are met. CARB has a requirement that the scientific underpinnings of all its regulations undergo scientific peer review. This is normally done by the University of California. Underlying this science-based approach is the willingness to move ahead in the face of some uncertainties.

*Air quality management plans and programs in California.* In the post-World War II boom period, California developed severe air quality problems. By the mid-1960s, total oxidant (ozone plus $NO_2$) levels approached 800 ppb in Los Angeles, and 24-h-average $PM_{10}$ concentrations exceeded 1800 $\mu g/m^3$ in desert areas and 600 $\mu g/m^3$ in Los Angeles. Although California made significant progress by attaining air quality standards for lead, $SO_2$, sulfates, and $NO_2$, and reducing peak ozone levels and PM, there are still many days of unacceptable ozone and particle levels across most of the state. In fact, over 90% of Californians continue to breathe unhealthy air at times.

Mobile sources such as gasoline-fueled vehicles (24 million cars and light trucks for 34.5 million people) and diesel-powered vehicles (1.25 million trucks and buses) play a major role in California's air quality problems. Because of California's proximity to the Pacific Ocean and geography, the meteorology is particularly conducive to generating poor air quality. Los Angeles' pollutant formation potential is the worst in the United States due to its unique combination of recirculation patterns, stagnation, inversions, and topography. The Los Angeles Air Basin's carrying capacity (an estimate of the maximum atmospheric burden a region can have and still attain air quality standards) per capita is five times less than Houston's (36 versus 181 lb VOC and $NO_x$/person/yr), which has similar ozone peaks. As a result of the state's poor air quality and large population, California residents receive more than 40% of the nation's population-weighted exposure to ozone values above the national 8-h standard of 0.08 ppm, and more than 60% of the population-weighted exposure to $PM_{2.5}$ values above the annual standard of 15 $\mu g/m^3$.

California's $PM_{2.5}$ nonattainment areas are dominated by ammonium nitrate and carbonaceous species, derived primarily from mobile sources. Unlike the East Coast of the United States and eastern Canada, California has greatly reduced sulfate levels. This is due to essentially removing sulfur from diesel fuel and gasoline, and the use of natural gas for electrical generation.

PM is California's greatest challenge, as it is responsible for over 6500 premature deaths per year (about 10 times greater than ozone and 20 times greater than cancer cases from known toxic air contaminants). Air pollution is estimated to cost Californians $51 billion per year—$4 billion per year in direct medical costs, with the remainder the value assigned to premature death. CARB calculates that California gains $3 in health benefits for every $1 it currently invests in air pollution control.

The concept of environmental justice (EJ), which entails the recognition that people of all races and incomes need equal protection from the detrimental effects of pollution, has emerged as an important issue in California over the past 5 yr. The debate focuses on the need for community controls in addition to statewide measures. In California, people who live near busy roads are disproportionately Hispanic, Asian, and Black, and from low-income families. Several Dutch studies found reduced lung function and higher asthma, hay fever, and wheezing rates for children living near heavy truck traffic (Brunekreef et al., 1997; Janssen et al., 2003). A study by

Ralph Delfino found that Hispanic children with asthma symptoms had higher breath levels of benzene, a marker for traffic (Delfino et al., 2003).

California is also concerned about indoor sources of air pollution. Kirk Smith of the University of California, Berkeley has calculated that a typical pollutant release is a thousand times, more likely to go down someone's throat if it occurs indoors rather than outdoors because people are usually indoors, near the sources (Smith, 1988). While the sources and risk reduction measures are known, CARB and other agencies have very little authority in this area.

California has adopted many emission standards more stringent than the U.S. standards. This includes light- and medium-duty vehicles' exhaust and evaporative standards, handheld and non-handheld small off-road equipment, personal watercraft, in-board motors for boats, and portable engines. Planned regulations for light-duty vehicles include a parts replacement program and improvements to the Smog Check program (i.e., more vehicles to test only, loaded mode testing for gasoline trucks, evaporative emission control test to detect liquid leakers). For forklifts and other large spark-ignited equipment, CARB is working on lower emission standards for new equipment as well as in-use reductions through catalyst retrofits. For heavy-duty vehicles, CARB has a broad range of controls to reduce emissions from both new and in-use vehicles (i.e., on-board diagnostics [OBD], reduced idling, chip reflash, gasoline tanker vapor recovery, in-use inspections in EJ areas) and must go beyond those strategies to get additional reductions. For off-road compression ignition equipment, although California is preempted from controlling a significant proportion (~80%) of this equipment, it is a huge source of emissions and large reductions are needed. California will work with the U.S. EPA to establish more stringent nationwide standards for hydrocarbons (HC), $NO_x$, and PM from off-road compression ignition engines, and to implement in-use strategies to get additional reductions. For marine engines, California plans to get reductions from existing harbor craft through cleaner engines and fuels. For the ports, reductions from land-based port emissions are planned, including cargo handling equipment and locomotives, heavy trucks, and dredges. CARB will set standards for additives to control engine deposits.

California has a goal of reducing diesel PM by 75% during this decade and 85% by 2020. This is being achieved with new emission standards, cleaner fuels, retrofits of existing engines, and enforcement programs. CARB and the U.S. EPA have adopted new vehicle standards that reduce emissions by 90% beginning in 2007. CARB will require aftertreatment on every diesel source where it is technically feasible. Low-sulfur fuel is required, as well as cleaner fuels like CNG (compressed natural gas) and measures to reduce or eliminate idling. Enforcement programs are used to minimize the effects of tampering and wear, especially in environmental justice communities.

California considers greenhouse gases to be ozone and particle precursors and recognizes that climate change can affect urban air pollution. In 2004, CARB adopted regulations that reduce greenhouse gases emitted by passenger vehicles and light trucks, although this measure is being litigated by the automotive industry. Reductions in greenhouse gases on the order of 30% can be achieved for all vehicle types using technologies already deployed in production vehicles. The costs are on the order of a few hundred to a thousand dollars and are more than offset by reduced operating costs of up to $5000. Gas–electric hybrid vehicles and other technologies can achieve greater reductions.

California set the bar for U.S. EPA and European Union emission standards that are now being adopted by many developing countries, particularly in Asia. Most of the world's population benefits from the fact that over 70% of the vehicles worldwide must comply with cleaner emissions standards. These policies have resulted in significant emission reductions and air quality improvements over the years. At least 50% reductions have been achieved in both the stationary and mobile source emission categories over the past 20 yr, and emissions will continue their downward trend (see Figure 18). These emission reductions have been achieved despite a doubling in vehicle miles traveled and a 50% increase in population. California's economy grew by 75% despite the $10 billion cost per year for air pollution measures adopted since 1990.

Air pollution levels have improved dramatically. The health-based standards for lead, $NO_2$, $SO_2$, and sulfates have all been attained, CO is very close, and peak ozone levels have dropped 75% relative to levels in the mid-1960s. California has also had success with $PM_{10}$ and air toxics.

CARB's technology-forcing emission standards have resulted in major advancements in emission control technologies. Today's cleanest passenger car emits less than 1% of ozone precursor emissions compared to the emissions from a car produced in 1960. California's successful introduction of many emission control programs has served as the basis for many similar U.S. programs. Through decades of emission control success, these programs have significantly improved California's air quality, despite more than doubling the number of people and tripling the number of vehicles over the last four decades.

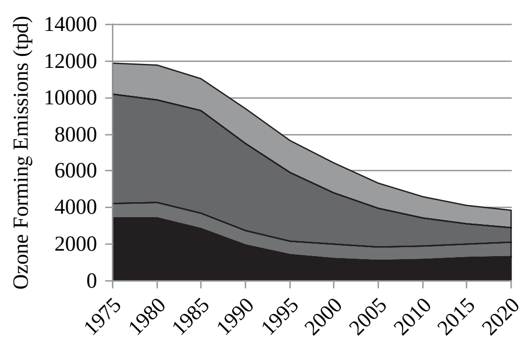



**FIG. 18.** Emissions reductions in both the stationary and mobile source emission categories over the past 20 yr. (CARB, 2002).

Two of the keys to CARB's success are the technical evaluations that go into its regulation development, and the transparent regulatory process. CARB develops new emission test methods, and in some cases proves that more stringent emission standards are achievable by funding or conducting technology demonstrations. It encourages participation by all stakeholders, including the public, industry, and communities that may be impacted by air pollution disproportionately from others. CARB meets with many stakeholders to hear concerns and to provide a mechanism for addressing their issues. It holds workshops that solicit suggestions and comments on initial issues. The technical data and assumptions are published in advance of the workshops. Regulations are first proposed in an initial report and additional workshops are held for public comment. CARB changes its proposal if significant issues are raised that warrant a revision. Once the regulation is adopted, it issues a formal response to all issues raised. The members of the public have a chance to air their concerns directly to CARB's board members, who are appointed by the governor and represent different professions and regions in California. The board reviews the technology and enforceability of regulations when necessary to make sure that the regulations meet the expectation held at the time of adoption.

Figure 19 shows a 29-yr timeline of the cost-effectiveness of various vehicle and fuel regulations, in dollars per pound of ozone precursor. Most measures have cost less than $2 per pound, which is considered to be quite reasonable in comparison to a benchmark of $5 per pound for stationary and area source control measures. Due to technology advancements, these costs have stayed fairly steady. CARB considers economic impacts of its regulations on California businesses and individuals, and regulations do not advantage or disadvantage California-manufactured products over products manufactured elsewhere in the United States or in the world. A recent study by EBI concluded that the air pollution control industry in California generated $6.2 billion in revenues and employed 32,000 people in 2001. The U.S. figures are $27 billion in revenues and employment of 178,000 people.

## Air Quality Management in Canada

*Historical Perspective on Air Quality Management in Canada*

In Canada, environmental management is an area of shared constitutional authority, with 14 governments (provincial, territorial, and federal) participating in air quality management activities. However, the main responsibility for controlling air pollution falls under provincial jurisdiction.

The federal and provincial governments first collaborated on air quality issues with the establishment of NAPS, the National Air Pollution Surveillance Network, in 1969, although the Canadian Department of the Environment wasn't created until 1971. The first federal regulation to address air quality was the Canadian Clean Air Act (CAA) in 1970, which limited release of chloralkali mercury releases from point sources beginning in the mid-1970s. The CAA Secondary Lead Smelter National Emission Standards Regulations soon followed, which restricted emissions of lead from secondary lead smelters.

In the mid-1970s, the first National Ambient Air Quality Objectives (NAAQOs) were developed by the federal government. These guidelines were nonbinding objectives, available for adoption by provinces as binding standards. Around the same time, vehicle emissions were regulated for the first time under the Motor Vehicle Safety Act. However, there was no further federal legislation covering air pollution until 1988.

In 1988, Canada passed the Canadian Environmental Protection Act (CEPA), which declared that "the protection of the environment is essential to the well-being of Canada." The act defined pollution control as a priority approach for

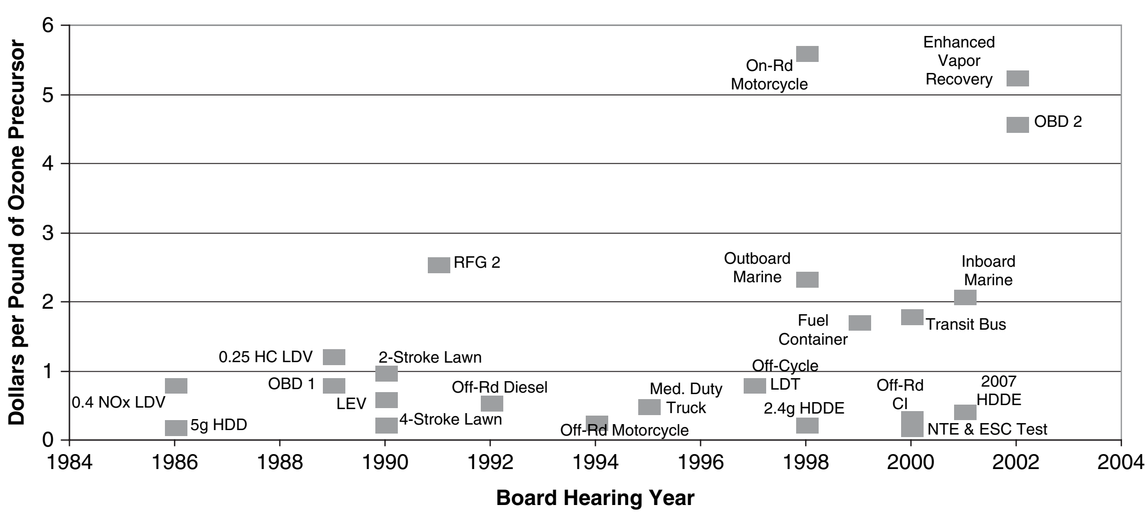



**FIG. 19.** Cost-effectiveness of various vehicle and fuel regulations, in dollars per pound of ozone precursor. (Reza Mahdavi, personal communication).

AQM and included several provisions which address air quality:

- Provisions to control the life cycle of toxic substances, including development, manufacturing, storage, transportation, use, and disposal.
- Regulation of fuels and components of fuels.
- Regulation of emissions from federal departments, boards, agencies, and crown corporations.
- Provisions to create guidelines and environmentally safe codes of practice.
- Provisions to control sources of air pollution in Canada where a violation of international agreements would otherwise occur.

In March 2005, CEPA 1999 was published after a 5-yr review process. The focus of CEPA 1999, which contains many amendments to the original act, is pollution prevention and the protection of the environment and human health in order to contribute to sustainable development. The updated act signified a departure from pollution control as a priority approach, setting deadlines for taking action to prevent pollution from toxic substances, and including the power to require prevention planning for toxic substances. CEPA 1999 also expanded the government's authority over fuels and engines, provided additional mechanisms for Canada to meet international obligations, and improved enforcement.

The development of AQM for smog in Canada began in 1990, when the federal and provincial governments committed to a national $NO_x$/VOC management program in what was intended to be the first of a three-phase plan. Designed to help Canada meet the Canadian maximum acceptable 1-h air quality objective for ozone of 82 ppb by the year 2005, it included over 60 initiatives with work shared among federal, provincial, and municipal governments. In 1993, the federal and provincial ministers of the environment and ministers of energy signed a Comprehensive Air Quality Management Plan, which was intended to coordinate federal and provincial initiatives on air quality. However, in 1997, they failed to reach consensus on phase two of the smog action plan, and the federal government developed its own Phase 2 Federal Smog Management Plan. The plan included several new initiatives for reducing $NO_x$ and VOCs, and identified PM as a contributor to smog. In the meantime, $PM_{10}$ limits were set by several provinces: Newfoundland, British Columbia (BC), and Ontario.

One approach to fostering intergovernmental cooperation on interjurisdictional issues such as air quality is through the Canadian Council of Ministers of the Environment (CCME). The CCME is an intergovernmental council composed of the 14 ministers of the environment for the federal, provincial, and territorial governments in Canada. The ministers meet twice a year to try to develop nationally consistent approaches to environmental management issues. However, the CCME does not have the power to enforce legislation; members of the council retain legislative authority for their own jurisdictions.

Initially, the CCME focused on individual areas of environmental protection, but in 1993, efforts to harmonize environmental programs and policies became a priority. In January 1998, the Canadian Environment Ministers (with the exception of Quebec) signed the Canada-Wide Accord on Environmental Harmonization. The objectives of harmonization are to (1) enhance environmental protection; (2) promote sustainable development; and (3) achieve greater effectiveness, efficiency, accountability, predictability, and clarity of environmental management for issues of Canada-wide interest.

Harmonization was intended to encourage cooperation among the provincial governments in development of consistent environmental measures such as policies, standards, objectives, legislation, and regulation across federal and provincial jurisdictions. The accord delineated the roles and responsibilities of federal, provincial, and territorial governments within a management partnership, so as to prevent overlapping activities and interjurisdictional disputes. A series of principles underlies the accord, including (1) polluter pays, (2) precautionary principle, (3) pollution prevention as a preferred approach, (4) performance-, results-, and science-based environmental measures, (5) transparency and participation, (6) cooperation with Aboriginal people, (7) flexible implementation, and (8) consensus-based decision making. The individual governments retain legislative authority and are not prevented from legislating stricter standards than those determined under harmonization.

One element of the accord was development of Canada-Wide Standards (CWS), intended to provide an alternative regulatory tool for the management of environmental issues of national interest. In 2000, federal and provincial governments (except for Quebec) endorsed CWS for ozone and PM with implementation target dates of 2010. Risk management approaches to air quality under CEPA now integrate both NAAQOs and CWS.

*Transboundary cooperation (based on Barton, 2008).* Canada and the United States are large countries separated by an international border that runs more than 8000 km, but they share airsheds and air quality issues. Much of Canada's population lives close to the international border with the United States, especially in southwestern Ontario. The prevailing airflow in this area brings air from the industrial areas in the American Midwest to Eastern Canada, and on smoggy summer days this can account for the majority of air pollution in Ontario and Nova Scotia.

In the 1980s, both countries began to experience the impacts of acid rain, which causes a cascade of damaging effects to lakes, streams, soils, aquatic wildlife, and vegetation. Areas experiencing acid deposition included the northeastern United States and parts of Ontario, Quebec, and the eastern Canadian provinces. In 1980, the two countries signed a memorandum of intent on transboundary air pollution, and in 1991, the Canada-United States Air Quality Agreement became the

first agreement on transboundary air pollution. Its Acid Rain Annex committed each country to specific emissions reductions: 10 million tonnes of $SO_2$ emissions nationally, including caps for power generation and industrial sources and 2 million tonnes of $NO_x$ reduction from power generation and vehicles in the United States, and in Canada, staged caps on $SO_x$ both nationally and in the eastern provinces, plus reduction in stationary source $NO_x$ emissions and implementation of a $NO_x$ control program for mobile sources. The agreement included a notification and consulting mechanism, compliance monitoring, and prevention of air quality deterioration and visibility protection. Although it was negotiated specifically for acid rain, it provided a useful framework for technical and scientific cooperation in general and required regular review, assessment, and public progress reports.

In 2000, Canada and the United States signed an Ozone Annex to the Canada–United States Air Quality Agreement, extending its scope to include transport of ground-level ozone between the two countries. The annex commits each country to reducing $NO_x$ and VOC emissions, precursors to ground level ozone and smog formation, with the United States focusing on summertime caps on industrial boiler $NO_x$ emissions, mobile source controls, and new source standards for $NO_x$ and VOC, while Canada focuses on an annual $NO_2$ power plant cap by 2007, improved fuel and engine regulations, and emissions from solvents, paints, and consumer products.

Cooperation between the two countries has resulted in significant reductions in acidic deposition, acid rain monitoring and ecological assessment programs, development of shared emissions inventories, and production of a joint transboundary ozone assessment report in 1999 and a joint transboundary PM science assessment report in 2004.

In 2003, Canada began working in cooperation with the United States to develop a Border Air Quality Strategy. Three pilot projects were undertaken to investigate approaches toward cooperative transboundary AQM:

1. The Great Lakes Basin Airshed Management Framework was undertaken in Southwestern Ontario/Southeast Michigan, an area where air quality standards for $PM_{2.5}$ and $O_3$ are routinely exceeded on both sides of the border. The goal of the project was to improve air quality coordination and information exchange between the two countries. The project increased understanding of the technical tools and information used in each country for assessing air quality, completed quality assurance for air quality on each side of the border, initiated research into health impacts of the area's air quality, and identified improved mechanisms for responding to cross-border complaints. The conclusions from the project were that coordinated AQM of an airshed spanning an international border is both feasible and desirable.
2. The Georgia Basin–Puget Sound International Airshed Project was undertaken in western Canada, in an area where past local AQM and public support have helped maintain a clean area where air quality standards are usually met. The goal of the project was to increase information exchange and knowledge of regional air quality issues that might have transboundary effects, and to explore policy options for better collaboration and management. Health research on effects of PM is underway, and ongoing collaboration is continuing to identify air pollution causes and solutions, and implement regional measures to reduce emissions from mobile sources, marine vessels, agricultural activities, and woodstoves.
3. The Emission Trading Feasibility Study was undertaken to investigate the possibility of developing a cross-border trading of capped $NO_x$ and $SO_x$ emissions. Environment Canada and the U.S. EPA reviewed the United States cap and trade program and modeled the environmental and economic benefits of cross-border trading, focusing mainly on electricity generators that burn fossil fuels and emit $NO_x$ and $SO_2$. The study was completed in 2006 and concluded that acid rain, smog, and regional haze are problems in both Canada and the United States, and would improve if caps comparable to United States levels were implemented in both countries. It also concluded that cross-border trading would not change the overall emissions reductions or the expected benefits to air quality and the environment, and that it would be cheaper to comply with the caps with trading than without trading. To be successful, cross-border trading would require that Canada have enforceable emissions caps for $SO_2$ and $NO_2$ with rigorous emissions monitoring and public reporting requirements, and that both countries make legislative and regulatory changes to give the allowances equivalency across the border, and a commitment to pursue implementation of the program.

The impacts of Canada's AQM strategies have been variable. Decreases of 27%, 6%, and 15% were observed for emissions of $SO_2$, $NO_x$, and VOCs between 1990 and 2000, but Canada's per capita emissions of VOCs remained high. $PM_{2.5}$ levels have dropped since the 1980s, but exposure to ground-level ozone increased between 1990 and 2003. In 2003, Environment Canada estimated that about 50% of the Canadian population resided in areas where ozone levels exceeded the 3-yr standard, and about 33% lived in areas where either $PM_{2.5}$ or both $PM_{2.5}$ and ozone were above the three year standards.

In 2007, the Federal Government introduced the Federal Regulatory Framework on Clean Air, which is intended to reduce air pollutants and greenhouse gases together.

### *Regulation of Air Pollutants in Canada*

The Content of this section is drawn from Raizenne (2003).

The regulation of pollutants in Canada occurs at the federal, provincial, and municipal levels. Although the federal government can sign international treaties on air quality, it must have permission from the provinces to take action.

*Canadian Environmental Protection Act (CEPA).* Under CEPA, the Minister of Health and the Minister of the Environment are responsible for developing a list of substances considered "toxic," and proposing at least one instrument to prevent or control those substances. These instruments may include regulations, pollution prevention plans, environmental emergency plans or environmental codes of practice. In 2001, the federal government declared $PM_{10}$ to be toxic, with an emphasis on the finer fraction $PM_{2.5}$. In 2003, ozone and the primary precursor pollutants to the secondary formation of smog, including $SO_2$, $NO_x$, some VOCs, and $NH_3$, were also declared to be toxic on the basis of their contribution to smog.

Under CEPA, the federal government also sets NAAQOs (National Ambient Air Quality Objectives). These are national targets intended to protect public health, the environment, and aesthetic properties of the environment. They take a long-term risk reduction approach to protecting the environment and public health while recognizing economic and technical limits. The targets, which were developed for $SO_2$, $NO_2$, CO, $O_3$, and TSP, are intended to provide background information, a uniform scale for assessing air quality in Canada, and to provide guidance to governments. Provincial governments are responsible for implementing air quality standards, but are free to design their own implementation plans—this may include adopting NAAQOs as enforceable standards. The targets were originally developed as a three-tier system, where three ranges of air quality (maximum desirable, acceptable, and tolerable) were identified. However, a 1992 review of the NAAQOs recognized evidence that many of the pollutants did not have identified effect thresholds, which is problematic for establishing scientifically defensible threshold values for air quality management. In response, the new design for NAAQOs was a two-tier system. It included a reference level, above which there are demonstrated effects on the health and/or the environment, and an air quality objective (AQO), a concentration that reflects a specified level of protection while acknowledging technical feasibility issues. In all cases the NAAQOs are to be effects-based values developed after an extensive scientific review of the evidence. Individual provinces are free to adopt the values as objectives or as an enforceable standard.

The two-tier system has not yet been formalized for the NAAQOs (although CWS have been developed for both $PM_{2.5}$ and $O_3$). As of 2001, annual publications by NAPS reporting on current air quality compare ambient levels of $SO_2$, $NO_2$, CO, and $O_3$ to the NAAQS (National Ambient Air Quality Standards), which were adopted in the United States.

The Canadian federal government, in consultation with provincial ministers of the environment, recently developed a set of ambient air quality standards known as Canada-wide standards (CWS). The CWS establish numeric targets for ambient $PM_{2.5}$ (30 $\mu g/m^3$ using a 24-h averaging time) and $O_3$ concentrations (65 ppb over an 8-h averaging time) that should be met by 2010. The federal, provincial, and territorial governments must publish implementation plans and support strategies of pollution prevention, continuous improvement, and keeping clean areas clean where ambient concentrations are below the CWS. Jurisdictions commit to establishing and maintaining ozone and PM monitoring networks, and designing management plans. Additionally, they commit to providing regular reports on progress and participation in reviews of the standards. Under the CWS, the federal government agrees to aggressively pursue reductions in transboundary flow of pollutants and precursors in areas where jurisdictional action alone will not be sufficient to meet the CWS. The federal government is also responsible for preparing PM and ozone guidance documents, reviewing jurisdictional implementation plans, and overseeing joint initial actions (i.e., multipollutant emissions reduction strategies, alternative transportation, and health and science updates).

The CWS recognizes that in many areas of Canada, ambient levels are already below the numerical targets, and provides for AQM in these areas through the principles of "Keeping Clean Areas Clean"—which recognizes that polluting "up to a limit" is not acceptable, and encourages pollution prevention and best management practices, and "Continuous Improvements"—which suggests that all jurisdictions should take remedial and preventive action to reduce emissions where practical, even if they are meeting the numerical CWS target.

In 2004, the Canadian government released the Federal Agenda on the Reduction of Emissions of Volatile Organic Compounds from Consumer and Commercial Products, which describes actions to be implemented between 2004 and 2010 and sets VOC emissions standards for products such as consumer products, automobile refinish coatings, architectural and industrial maintenance coatings, and pesticide products.

*Emissions regulations.* The Canadian federal government has responsibility for setting new equipment emission standards; however, provinces are free to set their own more stringent standards.

Up until 2000, emissions limits for Canada's on-road vehicles were promulgated under the Motor Vehicle Safety Act; however in 2000, this authority was transferred to CEPA. Under CEPA 1999, the On-road Vehicle and Engine Emission Regulations were promulgated. The regulations came into effect on January 1, 2004, and continued previous approaches toward harmonizing Canada's vehicle emissions standards with those of the United States. Vehicle and engine certification requirements are now aligned with those of the United States federal U.S. EPA requirements, including the U.S. Tier 2 program for new light-duty vehicles, light-duty trucks, and medium-duty passenger vehicles, as well as the U.S. Phase 1 and Phase 2 programs for new heavy-duty vehicles and engines. Vehicles and engines meeting the more stringent requirements will be phased in over the 2004–2010 model year period, with exact phase-in dates depending on the specific vehicle class. More stringent exhaust emissions standards for LDVs affect $NO_x$, nonmethane organic gases, CO, formaldehyde, and PM. The standards apply equally to all weight categories within the LDV category.

Phased emissions standards for HDVs affect mainly $NO_x$ and nonmethane hydrocarbons.

No regulations were in place to control emissions from off-road engines until December 2000, when the Ozone Annex to the 1991 Canada–United States Air Quality Agreement was promulgated. Canada committed to establishing regulations under CEPA 1999 that would align with federal U.S. EPA regulations. In 2000, before promulgation of the regulations, Canada signed memoranda of understanding (MOUs) with 13 engine manufacturers who agreed to supply engines that would meet U.S. EPA's Tier 1 standards. In 2005, the Off-Road Compression–Ignition Engine Emission Regulations were promulgated, introducing emissions standards for model year 2006 and later for diesel engines used in off-road applications. The regulations currently follow the U.S. EPA's Tier 2 and Tier 3 process with some minor differences, and it is anticipated that they will align with Tier 4 regulations in the future.

*Cleaner fuels regulations.* *Sulfur in Gasoline Regulations* published under CEPA in 1999 limit sulfur in gasoline to an average level of 30 mg/kg with a never-to-be-exceeded maximum of 80 mg/kg. They prescribed the phase-in of low sulfur in Canada in two stages: by July 1, 2002, a limit of 150 mg/kg was imposed, and by January 1, 2005, a limit of 30 mg/kg was in effect. The regulations also include never-to-be-exceeded limits of 300 mg/kg during 2004 and 80 mg/kg thereafter.

*Sulfur in Diesel Fuel Regulations* were passed in 2002 and stipulate strict new emissions standards. The maximum allowable limit for sulfur for on-road diesel was reduced to 15 ppm in June 2006. Beginning in 2007, off-road diesel fuel was limited to less than 500 mg/kg, and must reach a limit of 15 mg/kg by 2010. Locomotive and marine diesel fuels will be required to have less than 500 mg/kg of sulfur starting in 2007 but will have until 2012 before dropping further to less than 15 mg/kg.

Other fuels regulations limit the concentration of benzene in gasoline to 1.0% by volume and the concentration of lead in gasoline to 5 mg/L (30 mg/L for leaded gasoline). Use of leaded gasoline in motor vehicles has been prohibited in Canada since 1990.

In 1969, Canada established the NAPS Network, the National Air Pollution Surveillance Network, as a joint action of the federal and provincial governments. It has expanded to include monitoring of $SO_2$, CO, $NO_2$, $O_3$, and particulate at over 152 monitoring stations in 55 urban centers. Additional monitoring of VOCs, $NO_x$, and rural $O_3$ is carried out in support of the federal smog plan.

Individual provinces develop their own approaches to AQM, and may impose stricter regulation than what is required federally. For example, Ontario has its own anti-smog action plan, a collaborative effort initiated in 1995, as well as a provincial "drive clean" vehicle emissions testing program. The province is implementing new regulations to apply $NO_x$ and $SO_2$ limits in five new sectors: iron and steel, cement, petroleum refining, pulp and paper, and carbon black. The limits (which already exist for the power generation and nonferrous smelting sectors) will become progressively stricter over time. New standards are being introduced for 29 pollutants, several of which have not previously been subject to standards, and air dispersion models are being updated.

Many industries are also taking voluntary steps to reduce emissions, while the nongovernmental organizations continue to lobby government, industry, and the public to adopt practices that will reduce emissions levels. Many Canadian municipalities have also taken on a diverse array of local air quality management initiatives.

### *Air Quality Management Plans and Programs: Toronto Case Study*

The City of Toronto is located on the North Shore of Lake Ontario in the province of Ontario. It is Canada's largest city and one of the fastest-growing metropolitan areas in North America. The city itself has a population of ~2.5 million, and an average population density of ~4000 inhabitants/km$^2$. It is situated at the heart of the Greater Toronto Area (GTA), which encompasses 25 separate municipalities in an area of just over 7000 km$^2$. Ontario is Canada's most populated province, and in general, the population density in Ontario is highest along its Southern border, where the climate is mildest. Toronto is one of a string of municipalities extending from Windsor, in Southwest Ontario, to East of Kingston, and the population density across the region is much higher than in the province (or country) generally. The area is subject to wide variations in meteorology, which is affected by the nearby Great Lakes. In spring and summer, the cooler water in the lakes helps keep temperatures down, while in the fall and winter, moisture from the lakes increases precipitation and the latent heat of the lakes protects the region from cold.

Toronto is also vulnerable to long-range transport of pollutants from the United States. Monitoring data suggests that on average, only about 35–40% of PM in Toronto originates in the city (although higher relative contributions exist in some local areas), suggesting that 55–70% of PM may be transported into the city. Additionally, on days when smog levels are high in Toronto, about half of the ozone in the city is estimated to originate from outside Ontario. The other main contributor to air pollution in Toronto is the transportation sector. Recent modeling suggests that in Toronto, motor vehicles contributed about 20% of the PM in the city (Brook, 2007b). More broadly, the transportation sector was responsible for 50% of the $PM_{2.5}$, 63% of the $NO_x$, and 85% of the CO emitted in Ontario in 2001.

Actions at the provincial level have an important influence air quality in Toronto. The acid rain issue initiated Ontario's first actions on air quality. In the 1980s, the Government of Ontario and the base metals sector negotiated significant reductions in sulfur dioxide emissions. In 1985, the government introduced its "Countdown Acid Rain" program, which placed an annual $SO_2$ emission cap of 885 kilotonnes on Ontario, a reduction of 67%, to be attained by 1994. Regulations required specific reductions from four major emitters: Ontario Hydro fossil fuel power plants, the Inco nickel/copper smelter in Sudbury, the Falcon-bridge nickel/copper smelter in

Sudbury, and the Algoma iron ore sintering plant in Wawa. The emissions reductions began in 1986 and in the case of Ontario power facilities affected NO emissions as well. In 1997, the four individual regulations were consolidated into a single regulation that outlined the current requirements at the time. Most $SO_2$ in the province still originates from metallurgical industries such as copper smelters and iron and steel mills, as well as the coal-fired power generation facilities, petroleum refineries, and pulp and paper mills. The highest $SO_2$ concentrations are usually recorded in the vicinity of these industrial facilities.

Provincial actions on acid rain were complemented by actions at the federal level, including the 1991 signing of the Canada–United States Air Quality Agreement and its Acid Rain Annex, which imposed reductions on $SO_2$ and $NO_x$ emissions in both countries. The results of these activities on air quality in Toronto was significant: Overall concentrations of $SO_x$ and PM have declined in the city since the 1970s, as shown in Figure 20 and Figure 21. In Canada, emissions have dropped by 40% from 3.8 million metric tonnes to 2.3 million metric tonnes over the past 25 yr.

Actions on ozone have not resulted in the same remarkable improvements (see Figure 22). Like particles and acid rain precursors, ozone is subject to long-range transport. Up to 50% of the ozone in Toronto on smoggy days originates in the United States. However, ozone generated in Ontario affects other regions, traveling onward to Quebec, the Maritimes, New York, and New England. Between 1979 and 1997, average ozone concentrations in Ontario increased by 19%, with some regions experiencing maximum 1-h average concentrations as high as 140 ppb in 1998. However, there appears to be a decreasing trend in the province for maximum ozone concentrations of about 13% between 1980 and 2004 (Environmental Monitoring and Reporting Branch, 2006).

Beginning in 1993, the provincial ministry issued air quality advisories when air quality was poor. In 2000, the Smog Alert Program, operated jointly by the Ontario Ministry of the Environment and Environment Canada, was initiated. Warnings are issued to the public when smog levels are predicted to be persistent and widespread within the next 24 h, or when high levels of unexpected smog occur. The program operated for health regions in southern, eastern, and central Ontario, and issues a separate air quality index and forecast for larger municipalities such as the City of Toronto. Since 1993, the number of smog alerts in Toronto has varied between 1–48, depending on the year.

In 2000, Canada and the United States added the Ozone Annex to the Canada–United States Air Quality Agreement, committing both countries to specific objectives for volatile organic compounds and nitrogen oxides which will reduce transboundary flows of tropospheric ozone and its precursors. The Ozone Annex established a Pollutant Emission Management Area (PEMA), which includes central and southern Ontario, southern Quebec, 18 U.S. states, and the District of Columbia. The provinces and states within the PEMA region are the areas of primary concern for the impact of transboundary ozone. In 2002, Canada and the United States met the first requirement of the Annex, which was collection of monitoring data for ozone, $NO_x$, and VOCs from stations within 500 km of the international border.

Both $NO_x$ and VOC emissions contribute to smog formation. In Ontario, most VOC emissions originate with the transportation sector or from general solvent use. Most $NO_x$ originates with the transportation sector. Recently, studies have

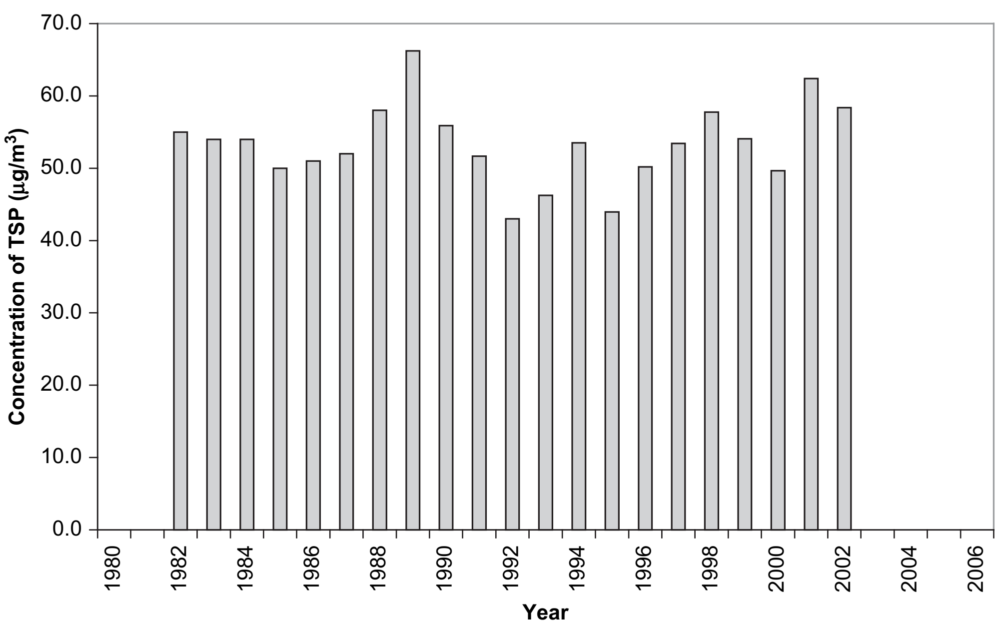



**FIG. 20.** Annual average (downtown) TSP concentration in Toronto. Image courtesy of Toronto Public Health; based on data from Ontario Ministry of the Environment Annual Reports.

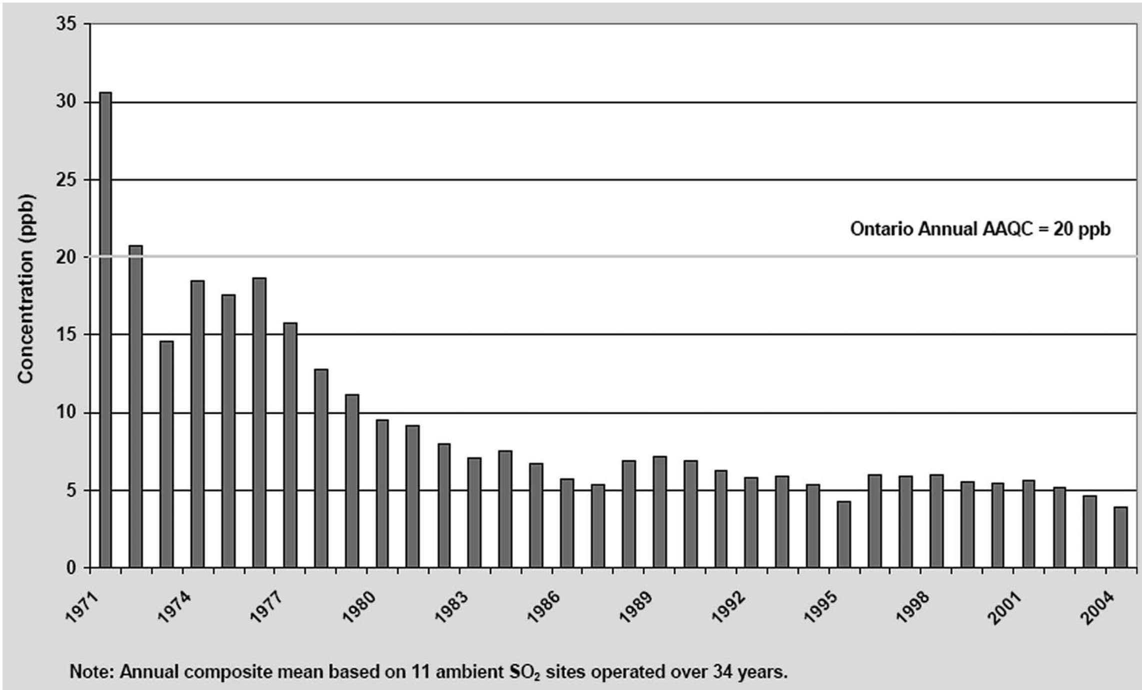



**FIG. 21.** 34 yr trend of sulphur dioxide concentrations in Ontario. (Environmental Monitoring and Reporting Branch, 2006). © Queen's Printer for Ontario, 2004. Reproduced with permission.

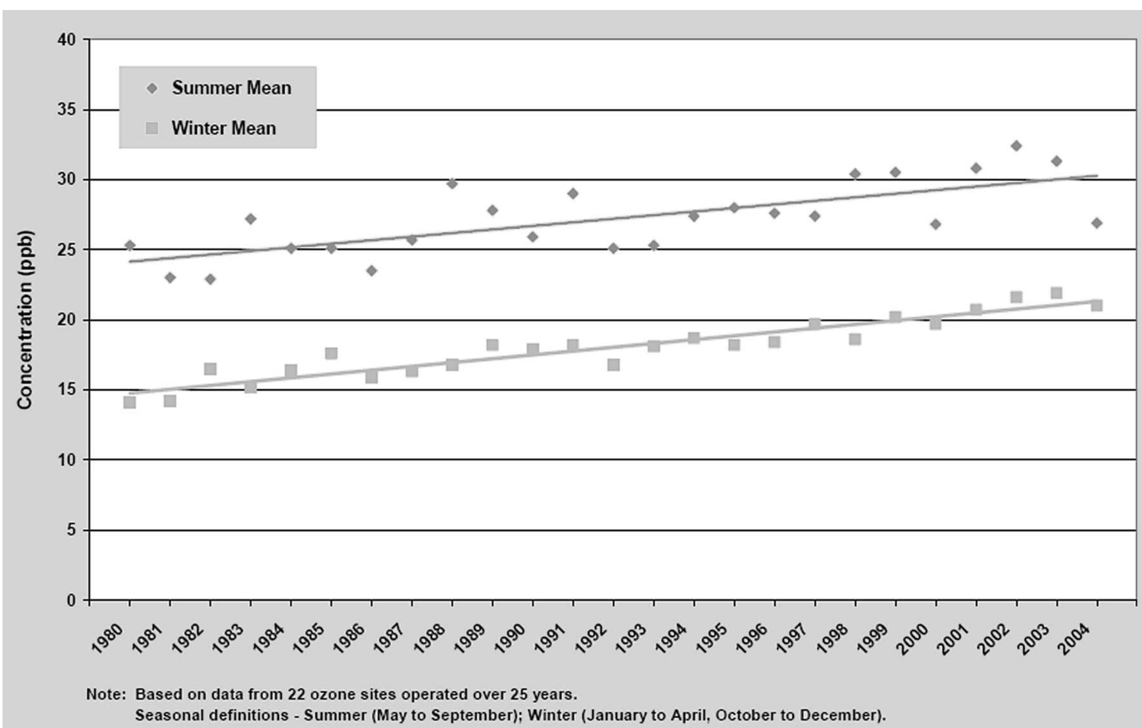



**FIG. 22.** Trend of ozone seasonal means at sites across Ontario (1980–2004). (Environmental Monitoring and Reporting Branch, 2006). © Queen's Printer for Ontario, 2004. Reproduced with permission.

shown that concentrations of $NO_2$ in Toronto are highest close to major highways and in the downtown core (Jerrett et al., 2007b). Traffic density, proximity to highways, and industry were all correlated to $NO_2$ in the city. Additionally, exposure to traffic appears to be correlated with increased rates of circulatory disease hospitalization and mortality in the city (Finkelstein et al., 2004). Concentrations of $NO_x$ have not changed dramatically since the 1970s (Figure 23).

In 1998, the city passed an anti-idling bylaw that limits idling to no more than 3 minutes out of every hour. The same year, the city initiated a corporate smog response plan, where activities such as reduced air conditioning use, evening refueling of vehicles, suspension of nonessential vehicle use, gas-powered lawn-mowing equipment, and use of oil-based products are curtailed on smog alert days.

In 2001, the provincial government introduced an emissions trading program for NO and $SO_2$, setting limits on the amounts permitted to be released from fossil-fuel-generating stations. However, critics of the emissions trading program suggest that because the government has committed to closing the power plants, which are the major source of the pollutants, the emissions trading system may not be an efficient way of addressing air quality issues.

The provincial government initially committed to closing its coal-fired power plants by 2007. These plants provide about 25% of the electricity used in Ontario, and will be replaced by a combination of several measures: reduced energy consumption through energy efficiency and conservation, investment in renewable sources, construction of new natural gas plants, and restarting of nuclear facilities that were closed in 1997. The initial phase-out date has been extended several times and now stands at 2014. The first plant to close was the Lakeview plant in 2005, which was previously the greatest source of air pollution in the GTA. Another coal-fired power plant, the Nanticoke generating station, has been labeled as Canada's worst polluter and is a major contributor to air pollution in Toronto.

*Burden of illness report and subsequent initiatives.* In 2000, the first Toronto Burden of Illness (BOI) report was published (Toronto Public Health, 2000), predicting that 1000 premature deaths and 5500 hospitalizations were caused in Toronto each year as a result of poor air quality. The report suggested that the impacts were preventable, and that air pollution increases severity or frequency of common medical conditions and illnesses. A follow-up BOI report published in 2004 predicted that 1700 premature deaths and 6000 result from air pollution in the city each year (Pengelly & Sommerfreund, 2004). The study was based on the health risk associated with acute exposures to ozone, nitrogen dioxide, carbon monoxide, and sulfur dioxide, as well as the health risk associated with chronic exposure to $PM_{2.5}$. The study also notes that air pollution affects thousands of people with less serious chronic illnesses such as chronic bronchitis and asthma.

The first Burden of Illness report garnered significant public attention and concern; it prompted Toronto's first smog summit in 2000, gathering representatives from each of the municipalities in the greater Toronto area (GTA) and the provincial and federal governments to discuss air quality in the region. The summit is now an annual event at which each of the governments can report on local actions to improve air quality, express concerns, and describe major barriers to improving air

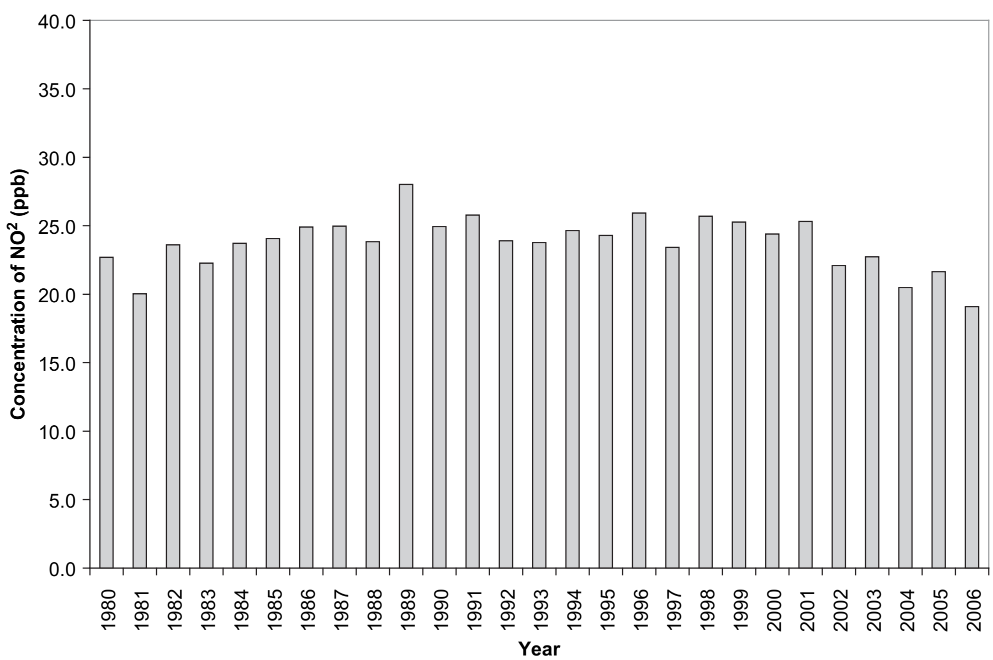



**FIG. 23.** Average annual $NO_2$ concentrations in Toronto 1980–2006. Image courtesy of Toronto Public Health; based on data from Ontario Ministry of the Environment Annual Reports.

quality. The summit has become an opportunity to increase knowledge and exchange ideas.

The BOI report also triggered the creation of *20/20 The Way to Clean Air*, a social marketing campaign designed and run by the public health units of York, Peel, Halton, Durham, and Toronto through the Clean Air Partnership (2005). The program emphasizes individual and collective actions that improve air quality and health in the city, setting a goal of a 20% reduction in energy consumption at home and on the road.

The report also instigated low-sulfur fuel purchases by the city, increased participation in policy discussions, and facilitated nongovernmental organizations (NGOs) in advocating for clean air. Finally, it promoted further research, giving rise to another report: *Condition Critical: Fixing our Smog Alert Warning System* (Toronto Public Health, 2001). This report suggested that the AQI (air quality index) in use did not adequately represent the risk to health from air pollution in the city, and did not appropriately warn the public about risks to their health. The AQI did not include particles, was based on air quality standards that are out of date, and was driven by the single pollutant with the highest concentration relative to its standard. Health Canada and Environment Canada have jointly developed a new AQHI (air quality health index) in collaboration with provincial, municipal and non-government organization (NGO) representatives. The AQHI has been piloted in several locations across Canada, including British Columbia and Halifax. Toronto was the first urban location to pilot the AQHI, beginning in July 2007. The other Greater Toronto Area (GTA) municipalities will join the AQHI pilot in the summer of 2008. Toronto Public Health has conducted a burden of illness study for air pollution from traffic in the City estimating 440 deaths and 1700 hospitalizations annually (Toronto Public Health, 2007). A number of initiatives are underway or are planned to reduce vehicle-related air pollution in Toronto.

Emission reduction initiatives at the local level play a critical role in air quality management. Municipal governments can contribute to cleaner air through emission reduction measures aimed at corporate fleets, energy conservation and efficiency measures in municipal buildings, public education to promote awareness and behavior change, transportation and land use planning, and bylaws (anti-idling etc). The Greater Toronto Clean Air Council has developed a resource to assist municipal governments in the development of an integrated approach to reducing air pollution in their communities. A Model Clean Air Plan for the Living City (www.cleanairpartnership.org/gtacac/pdf/clean_air_plan.pdf) provides a menu of options for measures that can be taken as part of a clean air plan and best-practice case studies of municipal clean air initiatives and lessons learned in their implementation. This Model Clean Air Plan seeks to reduce the quantity of fossil fuel that local governments and their communities use by reducing consumption, switching to cleaner fuels, improving energy efficiency, and reducing or phasing out activities or products that contribute to smog and climate change. It divides actions into five policy areas (transportation; energy; business, industry, and government; natural and built environment; education and outreach). The document notes that in general, many municipalities have found energy efficiency retrofits and improved energy management programs for their buildings, along with cleaner fuels procurement for their fleets, to be cost-effective ways to quickly reduce their smog-causing emissions, but municipalities will have to determine for themselves what is best for their particular situations.

The City of Toronto has been recognized as a leader in reducing greenhouse gas emissions and has recently released a new framework for an integrated climate change and clean air action plan (City of Toronto, 2007) to reach greenhouse gas and $NO_x$ and $PM_{10}$ emission reduction targets. The plan follows the policy trend in Canada and internationally toward an integrated and harmonized approach to cleaner air and lower greenhouse gas emissions. Twenty-seven proposed initiatives that residents, businesses, industry, and the Toronto government can take to reduce greenhouse gas emissions and create a sustainable urban environment are clustered according to four major energy sources that contribute to greenhouse gas emissions and poor air quality: natural gas, gasoline, diesel, and electricity. Some examples of proposed initiatives under each source are provided next:

1. Natural gas: energy efficiency retrofits for family homes, small businesses and high-rise dwellings; mandatory green building standards for new buildings, development of renewable energy systems on city-owned properties.
2. Gasoline: improved city transit plans, expand bikeway networks.
3. Diesel: convert city fleet to biodiesel, identify opportunities to replace food imports with locally produced goods.
4. Electricity: expand energy conservation and renewable energy conservation programs offered by local utilities, ban incandescent bulbs in city owned buildings, convert street lighting to LED, expand deep lake water cooling.

While there are many areas of critical need, climate change experts who provided guidance on the plan agreed that the transportation sector and the energy efficiency of residential and commercial buildings are crucial areas of opportunity.

### *Air Quality Management in Mexico*

This section is based largely on Molina and Molina (2002).

The Mexican air quality situation is dominated by the Mexico City Metropolitan Area (MCMA), home to 20 million people, 3.5 million vehicles, and 35,000 industries. The MCMA is thus the region of highest pollution in Mexico and the focus for air quality management activities. The activities initiated in the MCMA are beginning to spread to other metropolitan areas, and many of the policies adopted to address air quality problems in the city are national in scope.

### *Historical Perspective on Air Quality Management in Mexico*

*General environmental policy development.* Mexico's first attempts to create national regulations promoting appropriate use

of natural resources occurred with the "Air and Water Conservation Act of 1940." In 1972, the government created the Subsecretariat for Environmental Improvement in the Secretariat of Health and Care (*Secretaría de Salubridad y Asistencia*), acknowledging that health problems were arising from environmental pollution. Various laws were created to address environmental policy at a national level but there was little overall control of pollution and waste. It wasn't until creation of SEMARNAP (The Ministry of Environment, Natural Resources, and Fisheries) in 1994 that the regulations were coordinated by a single administrative body, as the need for a comprehensive plan that included economic, social and environmental considerations was acknowledged.

In 1995, a newly elected Mexican government paid special attention to environmental concerns for the first time. The two main objectives of its National Development Plan were to maintain economic growth and achieve sustainable development. A specific environmental strategy was developed with five focal areas:

- Link political instruments to the promotion and generation of employment and income.
- Ensure equitable distribution of costs and benefits, with the objective of fighting poverty.
- Reinforce preventive measures.
- Encourage social participation in policy design through mechanisms of consensus between social authorities and citizen groups.
- Actively participate in international forums and agreements.

In 2000, the Ministry responsible for environmental legislation changed names and became SEMARNAT (*Secretaría del Medio Ambiente y Recursos Naturales*), the Secretariat of Environment and Natural Resources, with a mandate to "create a State environmental protection policy reversing the tendencies of ecological deterioration and establishing the basis for sustainable development in the country." It set six main goals around which to structure its activities:

- Integrality.
- Commitment with all economic sectors.
- New environmental management.
- Assessment of natural resources.
- Observance of law and fight against environmental impunity.
- Social involvement and accountability.

Although SEMARNAT has principle responsibility for Mexico's environmental policy, some important enforcement duties are the responsibility of state and municipal governments.

*MCMA environmental policy development.* In the MCMA, a variety of air quality management plans have been implemented over the past few decades. Because air pollution from the city crosses jurisdictional boundaries, there has often been involvement from various levels of government. Construction of high-capacity roads and the expansion of the Collective Metro Transportation during 1978–1986 helped address congestion and air pollution to some extent, but initial air quality management plans such as PCMCA (*Programa Coordinado para Mejorar la Calidad del Aire en el Valle de México*), established in 1979, were largely unsuccessful.

A General Law passed in 1988 that assigned jurisdictional responsibility for environmental regulation encouraged development of new policy tools, and during the period from the mid 1980s to 1990, several important air pollution reduction measures were initiated, including:

- Conversion of roughly 2000 state-owned buses to new, low-emissions engines.
- Extending urban electric transit.
- Implementing no-driving day.
- Mandating a vehicle verification program.
- Developing and enforcement of a contingency plan for high-pollution days.
- Reduction of lead in gasoline sold in the MCMA.
- Gradual substitution of fuel oil with natural gas in the *Valle de México* power plant.
- Plans to move high-pollution industries out of the city.

At the end of the decade, the first comprehensive air quality management plan was developed for Mexico City. This was achieved as part of an extensive technical collaboration between the Federal District authorities, the World Bank, and the German Cooperation Agency.

Until the mid 1990s, environmental problems in Mexico and particularly in the MCMA were addressed through a "command-and-control" approach. This included use of official Mexican standards, and environmental licenses and reports of emissions from industrial facilities. More recently, an integrated approach including prevention, stakeholder input, training, technology transfer, and information dissemination has been favored. Since 1995, self-regulation instrument such as agreements between enterprise and government, voluntary standards, and environmental audits have also been adopted. Implementation of economic incentives such as taxes and subsidies is still rare.

The Comprehensive Program to Combat Atmospheric Pollution (PICCA) (*Programa Integral contra la Contaminación Atmosférica en la Zona Metropolitana de la Cuidad México*) was implemented in the MCMA from 1990 to 1995. Its goals were to reduce lead, $SO_2$, particulate, and $NO_x$ emissions, and the program oversaw a shift toward natural gas for industrial and power sectors, as well as reduction of lead in gasoline and sulfur content of diesel and fuel oils, introduction of two-way catalytic converters, the establishment of vehicle standards, and a no-driving program. The program was hampered by inefficiency and a lack of coordination among participating institutions.

PICCA was followed by a series of PROAIRE (*Programa para Mejorar la Calidad del Aire en el Valle de México*)

programs: air quality management programs for large urban centers in Mexico formulated with input from government, private, and public stakeholders. Their development and implementation is spearheaded by the INE (National Ecology Institute) with support from state and municipal authorities, academic institutions, NGOs, and the private sector. PROAIRE was first implemented in the MCMA, but has now been expanded to other regions of Mexico including the Guadalajara Metropolitan Area (1997–2001), the Monterrey Metropolitan Area (1997–2000), the Toluca Valley Metropolitan Area (1997–2000), Cd. Juárez (1998–2000), Tijuana-Rosarito (2000–2005), Mexicali (2000–2005), and Salamanca (2003–2006). These programs were implemented with funding from the U.S. EPA and the Western Governor's Association, with investigative support from MIT.

The goals of PROAIRE I included reduction of hydrocarbons, $NO_x$, particle emissions, and reduction of ozone peak and average concentrations in an effort to achieve greater compliance with guidelines. PROAIRE achieved introduction of MTBE into fuels, further reductions in sulfur content of fuels, reductions of aromatic content of gasoline, and implementation of Tier 1 vehicle emissions standards. Barriers to effective implementation of the program included lack of participation by all sectors, lack of coordination among the various participating institutions, and inadequate administrative and financial support.

The original PROAIRE was followed by a longer, more ambitious program, PROAIRE 2002–2010, which focuses on reduction of ozone and particulates. It includes a series of 89 individual measures targeting mobile, point, and area sources, and specifically addresses transportation and renewal of the automotive fleet. It is designed to improve the links between control options for urban air pollution and greenhouse gas emissions.

The PROAIRE program is intended to be a long-term policy initiative, but is subject to biennial reviews in which resources allocated to groups of policy measures are assessed, and new information is used to determine whether new measures should be added or existing measures abandoned.

Although emission inventories have been developed for the MCMA since 1986, the VOC to $NO_x$ ratios derived from the inventories do not reflect the ratios observed in ambient air, suggesting that the emissions models used were inaccurate. More recently, more reliable emissions inventories have been under development: between 1997 and 2000, the first emissions inventories were coordinated for the cities of Guadalajara, Monterrey, Ciudad Juarez, Tijuana, and Mexicali. As part of PROAIRE 2002–2010, the National Ecology Institute (INE) and SEMARNAT began developing a nationwide emissions inventory that includes point, area, biogenic, and mobile sources. The base year for the inventory is 1999, and it covers $NO_x$, $SO_x$, VOCs, CO, $NH_3$, $PM_{10}$, and $PM_{2.5}$. On September 18, 2006, Mexico released its first National Emissions Inventory (NEI), a tool that will inform ongoing institutional efforts to manage air quality.

Significant improvements in monitoring and evaluation of air quality have occurred in Mexico over time. Visibility range was the main indicator of air quality in Mexico from 1940 up until 1970. In 1940, the average visibility range was 4–10 km, in the 1950s it was 2–4 km, and it is now 1–2 km. In 1967, the Pan-American Network of Standardized Sampling was introduced, collecting $SO_2$ and TSP data at 14 stations, and in the 1970s Mexican authorities added an additional 22 manual $SO_2$ and TSP stations in collaboration with UNEP. In 1985, the U.S. EPA assisted with the installation of an automatic monitoring network, known as RAMA (*Red Automática de Monitoreo Atmosférico*). By 1999 there were 37 stations collecting data on $NO_x$, TSP, CO, and $O_3$, as well as data on $SO_2$ and Pb. In August 2003, the city government inaugurated a six-station $PM_{2.5}$ monitoring network now integrated by CENICA in a real-time system.

In the summer of 2006, the Registro de Emisiones y Transferencia de Contaminantes (RETC), Mexico's first mandatory pollutant release and transfer register (PTRER), was published, providing public access to detailed information about the release and transfer of 104 toxic chemicals.

### *Regulation of Air Pollutants in Mexico*

This material is drawn from Molina and Molina (2004a) and Molina et al. (2004).

Air quality was recognized as a social and environmental problem in Mexico beginning in the 1960s. The first law addressing air quality specifically, the Federal Law for Prevention and Control of Environmental Pollution, was passed in 1971, and in 1978, an Interministerial Commission for Environment was established to oversee the implementation of its regulations. A second national environmental legislation was the Federal Law of Environmental Protection, enacted in 1982 and amended in 1984 to include an air quality monitoring system. However, the new law had little effect since air quality guidelines and enforcement procedures were unaltered. Also, financial crises in the early part of the decade and the Mexico City earthquake of 1985 diverted attention and resources from the issue.

The primary legal mandate for air pollution prevention in Mexico at the national level is the 1988 General Law of Ecological Balance and Environmental Protection. The law assigns environmental responsibilities at various jurisdictional levels. The federal government has several responsibilities:

1. Issuing standards for air quality. This includes ambient air quality standards, maximum allowable emissions releases for industrial facilities, and emissions limits for vehicles. The current ambient air quality standards were adopted.
2. Permits for industrial facilities under federal jurisdiction (which includes most heavy industry, such as chemicals, energy, metals, cement, paper, cars, and transport). The government requires these facilities to install air pollution

control equipment, monitor emissions, and compile and submit emissions inventories.

3. Enforcement. The government may delegate some enforcement activities in agreement with state and municipal governments.
4. Air quality issues that affect multiple states.

The state and local governments are responsible for regulating light industry, vehicle use, including vehicle inspection and maintenance programs and driving policies, zoning, and measures to be taken under air quality "emergencies." They must also carry out air quality monitoring, and are responsible for developing transit plans.

The governance of the MCMA is split primarily between the Federal District (Distrito Federal or DF) and the State of Mexico (*Estado de Mexico* or EM). One of the major obstacles to the implementation of anti-pollution measures in the MCMA is the lack of a powerful metropolitan institutional structure. The Metropolitan Environmental Commission (*Comision Ambiental Metropolitana*, or CAM) was created in 1996 to coordinate the policies and programs that are implemented in the metropolitan area. Permanent members of CAM consist of the federal Secretariat of Environment and Natural Resources, the federal Secretariat of Health, the Chief of Government of the Federal District, and the Governor of the State of Mexico.

Every 2 yr, the responsibility to preside over CAM changes between the DF and the EM governments. Any decision on how to organize the Commission as well as the responsibility for operating costs would go to the jurisdiction in office at the time. Frequently, the side presiding over CAM has to use its own financial resources to manage the commission and its own environmental officials also serve as CAM officials. The local government that is not presiding over CAM, as well as the federal government, contributes human resources and other support to CAM operations, mainly for the specific tasks of its working groups.

The Environmental Trust Fund for the Valley of Mexico (*Fideicomiso Ambiental del Valle de Mexico*) was created exclusively to support CAM projects. Between 1995 and 1997, the Environmental Trust Fund received money collected from the application of a surcharge on gasoline sold in the MCMA. The annual renewal of the surcharge required the approval by the Finance Ministry, which did not happen in 1998. Since then, the surcharge has not been reactivated. The Environmental Trust Fund has its own organization and rules of operation, and it is managed through an Executive Committee headed by the Finance Ministry. One representative each from CAM, governments of the DF and the State of Mexico, and SEMARNAT are included. However, without income, the Trust Fund has been depleted. Other sources of funding for CAM projects include international environmental agencies, national and international financial institutions, international and national academic institutions, and foreign governments.

There are serious concerns over its current operation: One of the most important issues is that CAM does not have a specific budget for its own operation, nor does it have a defined operative organizational structure as well as lack of continuity. The Technical Secretariat is appointed by the presiding government, which rotates every 2 yr; in addition, local and federal representatives change in response to political events. These deficiencies in institutional memory cloud an integrated long-term vision of the policy requirements.

The Metropolitan Commission for Transport and Roadways (*Comision Metropolitana de Transporte y Vialidad*, or COMETRAVI) has a mandate similar to that of CAM, but it also lacks financial resources and has no executive or regulatory powers. In 1999, COMETRAVI developed a proposal for the adoption of comprehensive integrated strategies for transportation and air quality in the MCMA. This strategy has not been incorporated into the official programs.

The lack of integration of environmental policies with transportation, urban development, and land use planning is one of the most important barriers preventing sustainable environmental improvements. Another important barrier is the incomplete harmonization of environmental policies among the federal government, the State of Mexico, and the Federal District, which results in unfair practices and inefficiency. Also, at present neither local nor federal environmental agencies have sufficient human and financial resources to efficiently carry out their environmental management activities. Furthermore, the continuing dispersion and growth in the size of the MCMA drive the need for vehicle-miles traveled still higher. The almost totally unregulated establishment of communities on the periphery creates both mobility and environmental problems. The development of a regional planning commission with strong enforcement capability is fundamental to creating a sustainable transportation/ environmental system in the MCMA.

As a large source of emissions, the MCMA has the potential to influence air quality over a much wider region than the Valley of Mexico, thus exposing larger populations in nearby cities and also affecting forests and crops. Pollutants emitted outside of the MCMA likewise may influence air quality within the Valley of Mexico. Therefore, in addition to metropolitan coordination, there is an urgent need for regional coordination and planning. To ensure continuity in the implementation of long-term strategies, it is essential that the CAM be significantly restructured and be empowered to carry out the planning, integration, and implementation of metropolitan environmental policies.

*AQ indices.* Air quality for criteria air pollutants in the MCMA are reported as IMECA units (*Indice Metropolitano de Calidad del Aire*), or Metropolitan Index of Air Quality. They are derived from dividing the measured concentration of the pollutant by its concentration guideline. If the value of the ratio is less than 100, the air quality is considered satisfactory. Between 101 and 200 the air quality is considered unsatisfactory. Values between 201 and 300 indicate "bad" air quality, and 301–500

IMECA units indicates “very bad” air quality. If the IMECA units exceed a certain threshold (currently 240 for ozone, which is equivalent to 280 ppb), certain mitigation and adaptation measures are triggered: Activity of polluting industries is restricted, vehicle use is restricted, and outdoor activities at primary schools are curtailed.

The IMECA ozone threshold is higher than similar thresholds in the United States (about 205 ppb) and elsewhere.

*Case Study: Air Quality Management in Mexico City*

This discussion is based on Molina and Molina (2004b), with supporting evidence from MIT’s Laboratory for Energy and the Environment (2002).

The severity of the air quality problem in Mexico City has spurred large amounts of research and action, making it an ideal case study for learning about challenges facing those who undertake air quality research, management, and policy. Indeed, Mexico City serves as the case study for MIT’s Integrated Program on Urban, Regional and Global Air Pollution (http://mce2.org/airpollution/introduction.html), which was initiated in 1999 and has as its goal to “provide objective, balanced assessments of the causes and alternative cost-effective solutions to urban, regional and global air pollution problems through quality scientific, technological, social and economic analysis in the face of incomplete data and uncertainty” and will serve as a great resource for air quality management generally.

The Mexico City Metropolitan Area (MCMA) is one of the largest cities in the world and continues to expand rapidly. In 1950, the population hovered around 3 million and occupied about 120 km$^2$. Today, as a result of migration from other parts of the country and a rapidly industrializing economy, the city has swelled to almost 20 million inhabitants occupying 1500 km$^2$. An estimated 40 million litres of fuel is consumed per day in the MCMA (Molina & Molina, 2004a), generating smog precursors and pollutants. Located in an elevated basin at 2240 m above sea level and surrounded by mountain ridges on 3 sides, the city is prone to thermal inversions that trap pollutants in the MCMA basin. Pollution tends to be worst in the winter, when there is less rain and the inversions are more frequent; the high elevation and intense sunlight promote photochemical ozone formation year-round.

In 1992, the United Nations described Mexico City as the most polluted city on the planet. Between 1995 and 1999, the city’s population was consistently exposed to $PM_{10}$ concentrations above 50 μg/m$^3$ (the annual standard in Mexico), and 2 million MCMA residents experiences concentrations of 75 μg/m$^3$ or more. The daily maximum 1-h standard for ozone was exceeded 300 times per year or more (Mexico Air Quality Management Team, 2002).

Throughout the 1990s, successful reductions in concentrations of air pollutants such as lead, carbon monoxide, and sulfur dioxide were achieved as comprehensive air quality programs were developed and implemented, and monitoring and evaluation of air pollution improved.

In 1990, unleaded gasoline was introduced to Mexico City, and by 1997 was the only type of gasoline available in the MCMA. This has lead to a dramatic improvement in ambient lead levels (see Figure 24), with the guideline for lead not being exceeded since 1993. Additionally, blood lead levels in the MCMA population have declined.

Sulfur dioxide levels improved as heavy fuel oil and high-sulfur diesel was replaced by natural gas in industry and the power sector in the early 1990s. By the middle of the decade, heavy fuel oil use was completely phased out in the MCMA. In 1995, PEMEX, the state-owned oil company, replaced its high-sulfur diesel with a new variety containing 500 ppm sulfur.

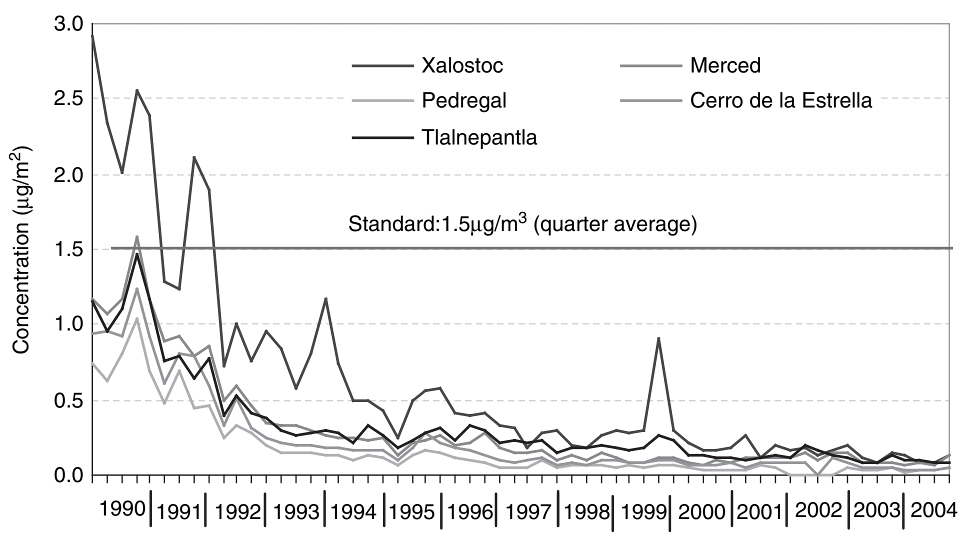



**FIG. 24.** Lead concentrations measured at five monitoring stations in the MCMA 1990–2002. (Bremauntz, 2008). Reprinted by permission of the publisher (Taylor & Francis Ltd.).

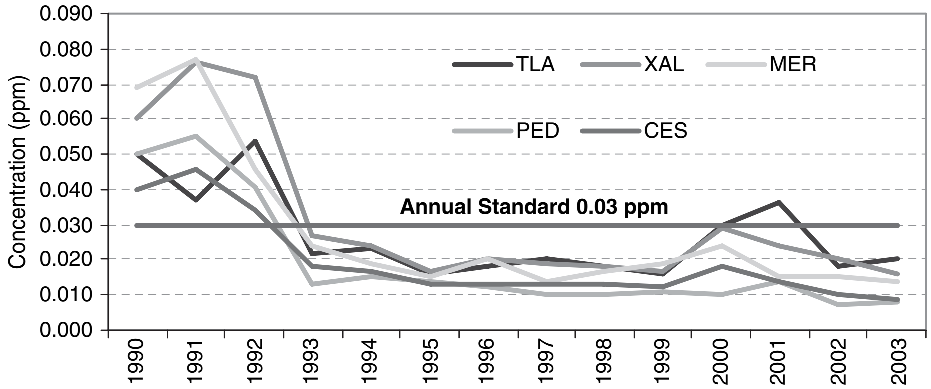



**FIG. 25.** Sulfur dioxide concentrations measured at five monitoring stations in the MCMA 1990–2002. (Bremauntz, 2008). Reprinted by permission of the publisher (Taylor & Francis Ltd.).



Figure 25 shows the dramatic decline in $SO_2$ concentrations across the MCMA in the mid 1990s. Emissions controls including membranes and floating roofs were installed in fuel storage tanks, and vapor recovery systems (phase 0, I, II) were incorporated into the gasoline system. Self-regulation schemes were promoted and fiscal incentives and duty tax exemptions were established to encourage cleaner technologies. Relocation of major industrial plants outside the valley and the 1990 closure of a large oil refinery near the MCMA also helped to improve air quality.

In recent years, a slight increase in sulfur concentrations has been observed, and is thought to be a function of illegal use of fuel oil by some industries in response to high natural gas prices.

*Transportation.* The transportation sector is a major contributor to Mexico City's air quality problems, contributing more than 99% of the CO, ~80% of nitrogen oxides ($NO_x$), 45% of the VOCs, and 80% of the $PM_{2.5}$. As the population has grown, it has also decentralized. Official estimates suggest that in 2020, a projected 37 million "trip segments" will be made in the MCMA daily, up from an estimated 29.1 million in 1994. In the past years, the percentage of trips made by fixed-route buses and on the Metro system has declined. The existing transit system has not adequately adapted to the changing population distribution and travel patterns, and low-income housing and new commercial developments have been built without adequate concomitant roadway construction or access to mass transit. This may be partly responsible for the shift toward increased vehicle ownership of about 6% per year. Use of *colectivos*, minibuses that follow a set route and stop frequently to pick up and drop off passengers along the way, has also increased. Emissions control equipment on these vehicles is usually not well maintained, and thousands of these minibuses compete for passengers in the city, increasing congestion. Taxis, many of which are older, inefficient vehicles, tend to have poorly maintained emissions control equipment, and contribute to congestion by driving around looking for passengers. Uncontrolled growth of both the *colectivo* and the taxi fleets pose difficult policy challenges.

The ageing freight fleet is also an important source of emissions in the MCMA, as goods are shipped to and around the city by truck, contributing to congestion and emissions. Due to a lack of routes that circumnavigate the MCMA, intracity freight often travels through the city.

Catalytic converters were first put into Mexican vehicles beginning in 1990, and by model year 1993, all cars were equipped with three-way catalytic converters. Stricter emissions limits were established, including adoption of U.S. Tier 1 standards in 1999, encouraging use of more advanced control technologies. At the same time, fuel quality was improving. As a result, emissions in the MCMA have decreased despite an increase in vehicle miles traveled. In 2000, the Mexican authorities reached an agreement with vehicle manufacturers for continuous improvement such that the Mexican standards attain equivalency with U.S. Tier 2 standards with a delay of 2 yr or less. At the same time, PEMEX has continued fuel improvement efforts and in 2006, introduced 50-ppm sulfur gasoline for the Tier 2 vehicles and 300-ppm vehicles for the rest.

In 1989, the government adopted the *Hoy no Circula*, or "no driving day" program, which imposed a rotating ban on personal vehicle use. Cars were prohibited from driving one day per week based on the last digit of their license plate number. The program, which grew out of a grass-roots initiative, was initially successful, and decreases in both pollution and congestion were observed. In the long term, however, it proved difficult to enforce: Many families bought a second car—often an older and more polluting vehicle. Ultimately, the program had the unintended effects of increasing driving by families that purchased a second vehicle and of drawing older, more polluting cars into the city from other regions of the country.

Beginning in 1993, the government strengthened and began to enforce the vehicle inspection program, which requires car owners to have their vehicles certified every 6 months, and harmonized procedures at Federal District and state inspection centers. In 1996, the inspection and maintenance was updated to act as an incentive to replace older, more polluting cars. Cars

aged 1993 or newer with catalytic converters are no longer subject to driving restrictions of any kind. Cars with electronic fuel injection systems but no catalytic converter (usually cars aged late 1980s to 1992) are banned from driving one day per week, but may be driven during pollution episodes that fall on permitted driving days. Cars with a carburetor and without a catalytic converter are subject to two restrictions: They cannot operate one day of each week or during any declared pollution episode.

Low-interest loans are being provided for vehicle substitution for *colectivos* and taxis made before 1992, and roadways and other infrastructure aimed at reducing congestion are being built. The government has created incentives to persuade companies to invest in cleaner vehicles, and has used subsidies to encourage manufacturers to furnish gasoline-powered delivery trucks with cleaner liquefied petroleum gas.

As a result of air quality management activities, the concentrations of CO, $NO_x$, and $SO_2$ (see Figures 25, 26, and 27) have also decreased and now rarely exceed the air quality guidelines. The most significant impacts are attributed to introduction of catalytic converters and improvement in fuel quality, and, to a lesser extent, to implementation of stricter industrial regulations and conversion to natural gas by the power plants. However, the ozone levels (see Figure 28) and PM levels still remain unacceptably high.

In response to increased vehicle traffic, the government is extending the metro lines and introducing bus rapid transit. The government is making efforts to improve service quality, facilitate transfers between modes of transportation, and improve personal security.

In 2002, Mexico City's environment secretary signed an agreement with the World Resources Institute (WRI) to create the Center for Sustainable Transport in Mexico City (CSTMC). The broad mission of the CSTMC is to devise a sustainable transport network for the city of 20 million, and its initial goals were to focus on bus rapid transit, engine/fuel

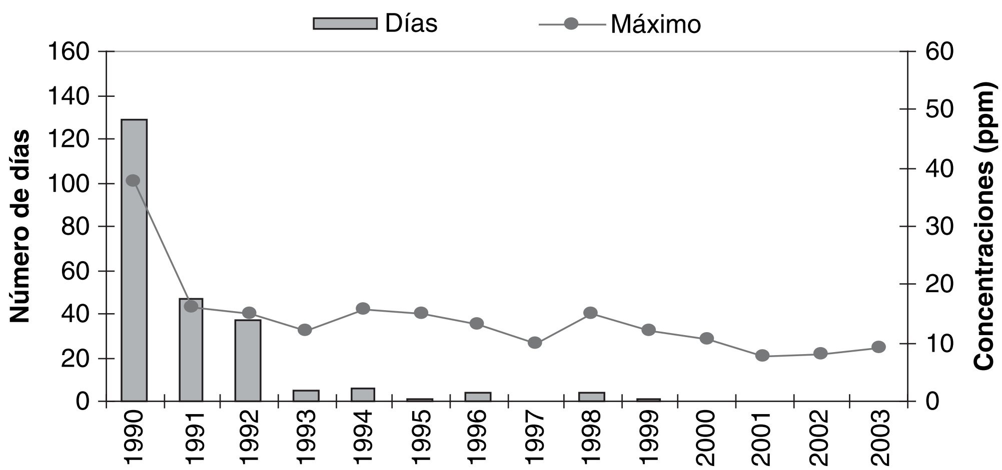



**FIG. 26.** Carbon Monoxide concentrations measured at five monitoring stations in the MCMA 1990–2003 (Bremauntz, 2008). Reprinted by permission of the publisher (Taylor & Francis Ltd.).

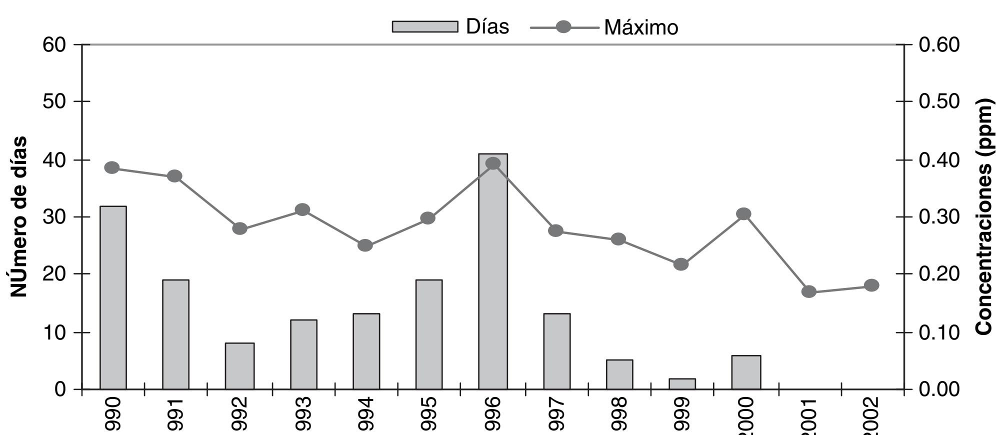



**FIG. 27.** Nitrogen dioxide concentrations measured in Mexico City: days above 1 hour standard (0.21 ppm) and 1 hr maximum concentrations. Data from 5 representative monitoring stations. (Bremauntz, 2008). Reprinted by permission of the publisher (Taylor & Francis Ltd.).

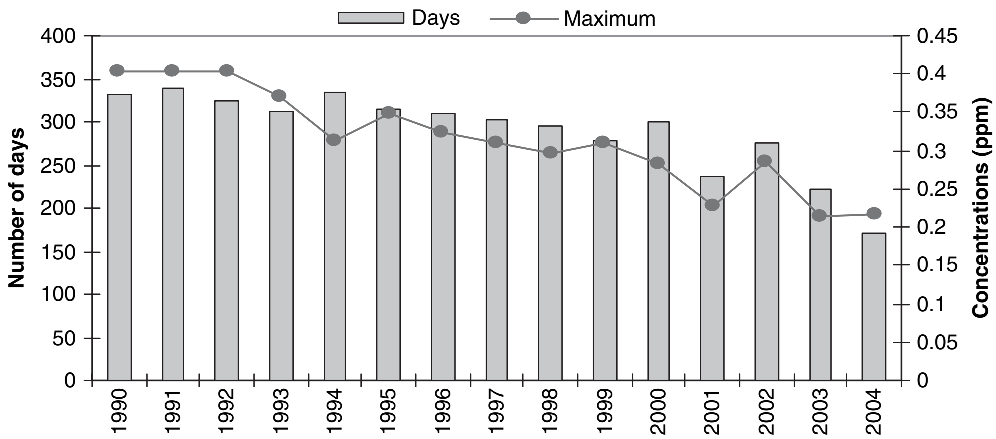



**FIG. 28.** Ozone concentrations measured in Mexico City: days above 1 hour standard (0.11 ppm) and 1 hr maximum concentrations. Data from 5 representative monitoring stations. (Bremauntz, 2008). Reprinted by permission of the publisher (Taylor & Francis Ltd.).



combinations for the high-capacity buses, and retrofitting existing diesel vehicles (Samaniego & Figueres, 2002). In June 2005, a Bus Rapid Transit system was launched, with designated bus lanes running along 14 km of the central transport artery Insurgentes Avenue. A recent pilot project to retrofit Mexico City's diesel buses with catalytic converters and diesel particulate filters, using ultra-low-sulfur diesel (ULSD) fuel imported from the United States, showed that particulate matter (soot) emissions could be reduced from older buses by 20–30% and from newer buses by up to 90%. Partly because of these findings, PEMEX will accelerate its plan to produce ultra-low-sulfur diesel, making it available by 2007.

Mexico has also instituted land use planning for both urban development and rural areas, ecological restoration, including rural and urban reforestation, human settlement control in rural areas, and environmental education and research, including establishment of an Epidemiological Surveillance System and increase in air quality research activities.

Various other initiatives that have been introduced for learning purposes include the pilot use of solar water heaters, introduction of efficient lighting on a massive scale, testing of electric vehicles, and a carbon sequestration project in the south of the Federal District.

*Benefits.* The impact of poor air quality on health is discussed more fully elsewhere in this document. However, the issue is particularly important for residents of the MCMA, where the higher altitude and concomitant decrease in oxygen means that more air (and pollutants) may be inhaled overall by residents in order to obtain sufficient oxygen. In 2002, Evans (Evans et al., 2002) estimated the benefits of a 10% reduction in PM and ozone at US$760,000 to US$2,200,000. In 2004, Molina and Molina reached a similar conclusion, estimating that a 10% reduction in PM and a 10% reduction in ozone would be associated with benefits of US$1,950,000, related to 3000 fewer deaths (due to PM reduction), and 300 fewer deaths annually (due to ozone reduction) (Molina & Molina, 2004a).

*Conclusions.* As a result of air quality management activities, the concentrations of CO, $NO_x$, and $SO_2$, have decreased and now rarely exceed the air quality guidelines. The most significant impacts are attributed to introduction of catalytic converters and improvement in fuel quality and, to a lesser extent, to implementation of stricter industrial regulations and conversion to natural gas by the power plants.

However, decreases in PM, ozone, and $NO_2$ are inadequate. The PM standard is violated on ~40% of days, and the ozone standard is violated on ~80% of days. Barriers to implementing air quality control measures include lack of financial resources, lack of information, and inadequate follow-up.

The benefits of improving air quality have been established and the government, with assistance from a variety of international organizations, is continuing to address air quality in Mexico, and specifically in the MCMA.

## Air Quality Management in the European Community

### *Historical Development of Air Quality Management in the European Community*

*European Union legislation on environmental issues and air pollution.* An important starting point for the development of environmental policy was the first United Nations Conference on the Environment in Stockholm in 1972. In 1972 the European Council made a commitment to establish a European Community environmental policy. The first so-called Environmental Action Programme (EAP, http://ec.europa.eu/environment/env-act5/envirpr.htm) was decided in November 1973 and laid down principles for the environmental policy in the community. It emphasized, inter alia, that economic development, prosperity, and the protection of the environment are mutually interdependent.

However, environmental considerations were also always linked to other considerations relevant for policy development

within the European Community, e.g., the setting of uniform emissions standards to avoid distortions to industry competitiveness. Product regulations had to be harmonized in order to avoid nontariff barriers originating from different national product norms. On the other hand, the economic benefits, especially the positive employment effects to be gained from environmental policies, were stressed.

Environment policy was built into the treaty by the Single European Act of 1987, and its scope was extended by the Treaty on European Union on 1992. This allowed the use of majority voting on environmental legislation. The general objectives formulated now in the treaty are to:

- Preserve, protect, and improve the quality of the environment.
- Protect human health.
- Utilize natural resources in a prudent and rational way.

For achieving these environmental objectives the treaty explicitly lists the precautionary principle, the principle of preventive action, the principle of rectifying damage at the source, and the polluter pays principle.

In 1992, the EC set itself the objective for achieving sustainable development. The long-term goal, to transform the European economy into one whose development would be sustainable for generations to come, was set out in the 5th Environmental Action Programme 'Toward Sustainability' (http://ec.europa.eu/environment/actionpr.htm). In addition, the 5th Environment Action Programme calls for the effective protection of all people against recognized health risks and demands that the guideline values of the World Health Organization (WHO, 2000) should become mandatory at the European Union (EU) level.

The 6th Environmental Action Programme (covering the period from 2001 to 2010) identifies four environmental areas for priority actions:

- Climate change.
- Nature and biodiversity.
- Environment and health and quality of life.
- Natural resources and waste.

The main avenues for action include:

- Effective implementation and enforcement of environmental legislation: necessary to set a common baseline for all EU countries.
- Integration of environmental concerns: Environmental problems have to be tackled at the source and frequently this requires the integration of environmental concerns into sectoral policies.
- Use of a blend of different approaches: All types of instruments have to be considered, not just legislation. The essential criteria being optimal efficiency and effectiveness.
- Promoting of participation and involvement across society—business, citizens, NGOs and social partners—through better access to quality information on the environment and cooperating to devise solutions.

In addition, the EAP requires the European Commission to prepare Thematic Strategies covering seven areas, including air pollution.

Environmental legislation leaves plenty of scope for national action and allows Member States to take tougher protection measures than those agreed at the EC level. The situation is different for legislation affecting the free movement of goods (e.g., product regulations). Stricter regulations may only be applied in special cases.

*Development of air quality legislation in the European Community.* The first so-called Directive of the European Community on air quality entered into force in 1980 (Directive 80/779/EEC). This directive set air quality limit values and guide values for sulfur dioxide and suspended particulates. The directive specified a date by which the limit values had to be attained, but also allowed for a prolonged period of noncompliance in zones if a Member State could show that plans for the progressive improvement of the quality of the air in those zones were developed. From today's perspective, the limit values were rather high. In 1982 and 1985, new directives on lead and nitrogen dioxide, respectively, entered into force. These directives also contained limit values.

In 1992, an ozone directive was decided. This directive did include certain thresholds for the assessment of air pollution and for the warning of the population, but did not request emission reductions in the case of exceedances of these assessment thresholds.

In 1996, the Environment Council adopted an Air Quality Framework Directive (FWD) 96/62/EC on ambient air quality assessment and management. This directive covers the revision of previously existing legislation, the introduction of new air quality standards for previously unregulated air pollutants, and setting the timetable for the development of daughter directives on a range of pollutants. The list of atmospheric pollutants to be considered includes sulfur dioxide, nitrogen dioxide, particulate matter, lead, and ozone—pollutants governed by already existing ambient air quality objectives—and benzene, carbon monoxide, polycyclic aromatic hydrocarbons, cadmium, arsenic, nickel, and mercury. The so-called daughter directives to the Air Quality FWD are described in more detail in an earlier section of this review.

The Air Quality FWD and its daughter directives are only one pillar of the EU air quality legislation. A number of other directives had considerable (and partly larger than the air quality directives) impact on air quality, notably those setting emission standards for mobile and stationary sources. In addition, some directives regulate product standards; a few of these are also particularly important for air quality (such as the directives on fuel quality, solvents, etc.).

The most important EC directives that impact air quality include:

- The EURO standards have established emission limit values for different pollutants for cars (differentiated between diesel- and gasoline-fueled) and light- and heavy-duty vehicles. As an example, the development of emission limit values (ELV) for cars ($NO_x$ and PM) is given in Figure 29. (However, it has to be noted that real-life emissions can be considerable higher than the ELV.)
- The directive on large combustion plants (LCP, 2001/80/EC) sets more or less stringent emission limit values for large installations in the power generation sector.
- The directive concerning integrated pollution prevention and control (IPPC, 1996/61/EC) requires the implementation of the best available technology (BAT) concept to a large number of industrial activities (energy industries, production and processing of metals, mineral and chemical industries, waste management, etc.), for which it lays down general rules for the national permitting systems. The directive covers both new and existing installations. The basic concept is that operators should go as far as they reasonably can to optimize their environmental performance by applying the best available techniques. Environmental performance is eventually to be measured against meeting the existing environmental quality standards, e.g., for air pollution to comply with the air quality standards of European Community legislation. Measures going beyond BAT may be requested if this is necessary to achieve EC environmental objectives. The IPPC directive covers only larger installations (>50 MW). However, there is no comparable EU legislation for small (including the domestic sector) and medium installations, even though these source categories may contribute significant to excess air pollution.

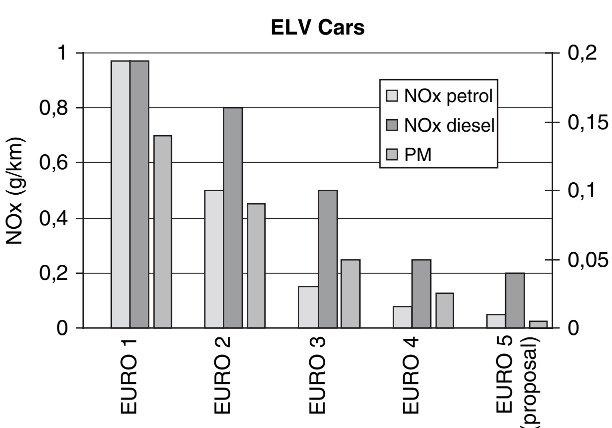



**FIG. 29.** Development of emission limit values (ELV) for cars in the EU for $NO_x$ and PM.

- The Directive on National Emission Ceilings (NEC, 2001/81/EC). This directive sets national emission ceilings for the pollutants $SO_2$, NOx, NMVOC, and $NH_3$.
- The Directive on Volatile Organic Compounds Emissions from Storage and Distribution of Petrol (94/63/EC).
- The Directive on Solvents Use in Industry (99/13/EC).
- The Directive on Sulphur Content of Liquid Fuels.
- The Directive on Emissions from Engines to be Installed in Non-Road Mobile Machinery.
- The Directive on the Quality of Petrol and Diesel Fuels.
- The directive on emission of VOCs due to use of organic solvent.
- The directive on the incineration of waste.

*Trends in Emissions in the European Union*

Even though there was continued economic growth in the past decades, emissions in general stabilized or decreased. As an example, the aggregated emissions of PM (primary and precursors for secondary PM) are shown in Figure 30.

This decoupling was triggered by stringent legislation, but also by other factors including fuel switching (which was partly influenced by economic considerations).

This is illustrated for the power sector and $SO_2$ emissions in Figure 31.

*Regulation of Air Pollutants in the European Union*

Numerical limit and target values as well as threshold values were set in four so-called daughter directives (DDs) to the Air Quality FWD. The limit values from the first and second DDs are listed in Table 5 (the third and fourth DDs only contain target values). The table also contains information on the basis for setting standards. As requested in the 5th EAP, many numerical values are identical to WHO AQG levels as contained on the Air Quality Guidelines for Europe (WHO, 2000). For some pollutants, the WHO AQG do not contain a numerical level (such as for PM and benzene). The respective limit values were based on recommendations from technical working groups.

There are some important specifics about the limit value (LV) concept of EU legislation. The most important include:

- The LVs have to be attained within a given period and are not to be exceeded once attained. This definition implies that the limit values are not a weak environmental objective, but a strict requirement (which in principle also constitutes individual rights for citizens).
- The limit values apply in principle everywhere (including hot-spot locations) except at workplaces (in reality, compliance monitoring and therefore compliance assessment include hot spots, but usually focus on those hot spots where exposure can occur).

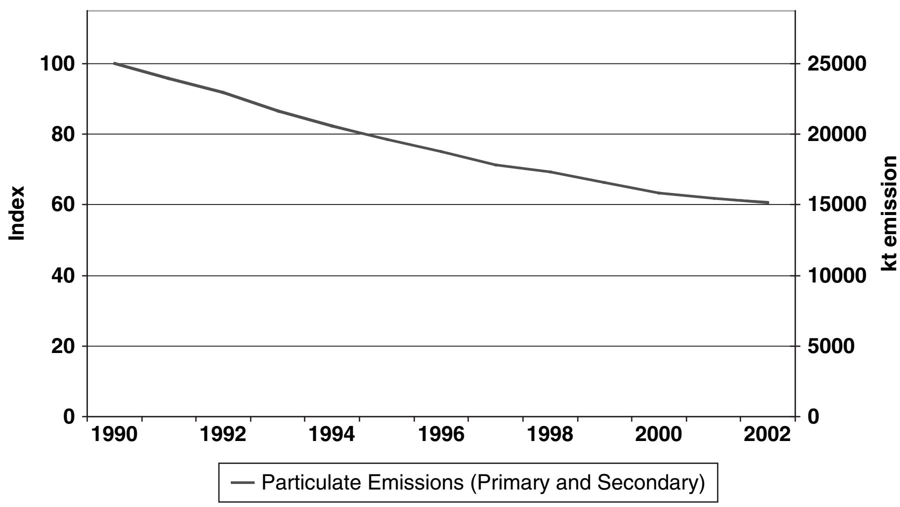



**FIG. 30.** Emissions of primary and secondary fine particles (EU-15), 1990–2002. (European Environment Agency, 2006b).

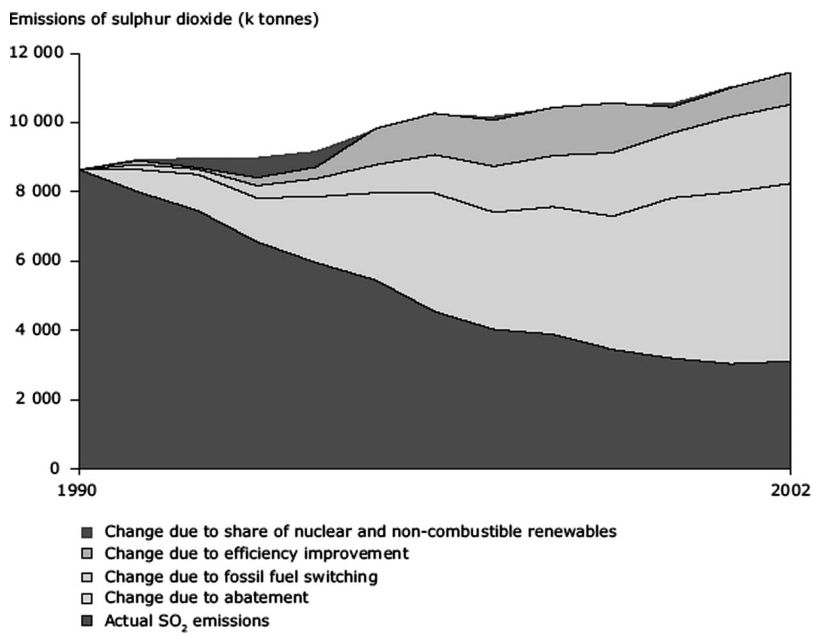



**FIG. 31.** Development of $SO_2$ emissions from the power generation sector in the EU 15. (European Environment Agency, 2005).

*Strategies for Enforcement of Regulatory Measures*

In general, the European Community has a relatively strict system to enforce implementation of European Community legislation.

EU legislation has to be transposed and implemented in EU Member States. Transposition and implementation is scrutinized by the European Commission, the "safeguard" of the treaty (and secondary legislation). If legislation is not implemented sufficiently, the European Commission may start a so-called infringement procedure, which has several steps. At the end, there is the possibility that Member States are condemned by the European Court of Justice, which can also result in considerable fines (which have to be paid by the Member State).

Noncompliance with limit values has also lead to national court cases in different Member States. In Austria, there has been a ruling by an appealing court implying that the authorities might be liable to damages compensation if there is health damage due to access air pollution.

In addition, licensing of new (usually quite clean) plants in areas with air quality in the range or above limit values is usually only possible if these installations have small contributions to air pollution or if their emissions are compensated by other measures. Therefore, there is often a clear interest by

**TABLE 5**
Numerical values of air quality limit values in the EU

| Pollutant | Averaging period | Limit value | Basis for setting standard | Exceedances allowed |
|---|---|---|---|---|
| $SO_2$ | 1 hour | 350 µg/m$^3$ | Based on WHO AQG level[1] | 24 |
| | 1 day | 125 µg/m$^3$ | WHO AQG level | 3 |
| $NO_2$ | 1 hour | 200 µg/m$^3$ | WHO AQG level | 18 |
| | 1 year | 40 µg/m$^3$ | WHO AQG level | – |
| $PM_{10}$ | 1 day | 50 µg/m$^3$ | Risk assessment in combination with an assessment of feasibility | 35 |
| | 1 year | 40 µg/m$^3$ | Risk assessment in combination with an assessment of feasibility | – |
| Benzene | 1 year | 5 µg/m$^3$ | Risk assessment in combination with an assessment of feasibility | – |
| Lead | 1 year | 0,5 µg/m$^3$ | WHO AQG level | – |
| CO | 8 hour | 10 mg/m$^3$ | WHO AQG level | – |

[1]WHO Air Quality Guidelines (AQG) for $SO_2$ is 500 µg/m$^3$ as 10-minute average.



industry to promote emission reductions in other sectors in order to avoid nontechnical barriers in licensing of new plants.

*Air Quality Management Plans and Programs in the European Union*

Stringent ambient air quality standards by themselves do not provide protection. The main tools to achieve the limit values are so-called plans and programmes (if the sum of the limit value and a so-called margin of tolerance is exceeded) and, after the attainment date, action plans, which have to be implemented if there is a danger of exceeding limit values. The limit values for $PM_{10}$ and $NO_2$ are rather stringent, and exceedances are frequent in some parts of Europe. This triggered the development and implementation of air quality plans to reduce pollution. These plans also have to be reported to the European Commission. The plans are currently scrutinized in a project funded by the European Commission. Figure 32 provides an overview of the pollutants covered by the plans and those sources that have been identified as main source of this pollutant.

There is some flexibility concerning the implementation in Member States, e.g., concerning responsibilities. There are large differences concerning the responsible authorities; in some Member States, local authorities are responsible for air quality assessment and management, while there are also examples where the responsibility lies with regional or national authorities. There is no simple answer to the question of which model is most effective.

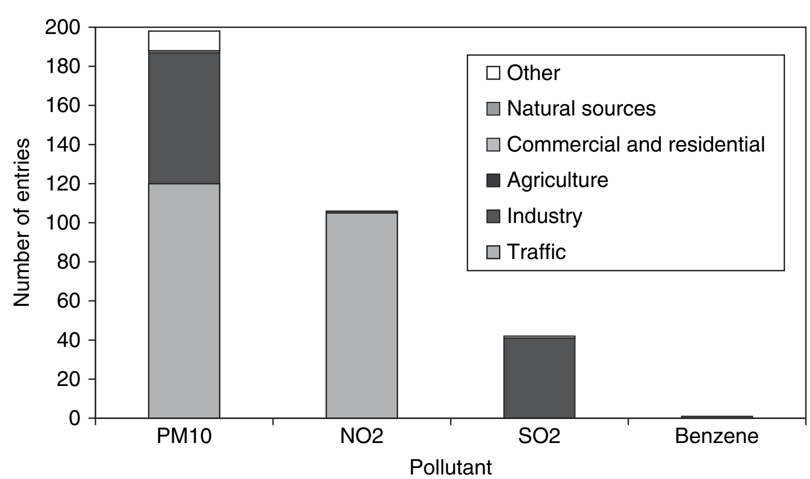



**FIG. 32.** Main sources listed for different pollutants in plans and programmes reported to the European Commission for exceedances of limit values between 2001 and 2003.

*Exposure reduction target (ERT).* Experience has shown that for nonthreshold pollutants, single limit values or standards may not on their own be the most appropriate way of managing air quality, particularly in areas where existing air quality management systems are mature. This has encouraged the European Commission to propose a new, additional concept, the exposure reduction target (ERT) (which has not entered into force yet, even though the concept is in principle supported by the European Council and the European Parliament). The following short description of the basics of the concept is derived from a non-paper issued by the European Commission.

The existing legal framework of the Air Quality FWD and its daughter directives requires complete compliance, meaning that limit values must be met everywhere continuously. As such, a conventional air quality management strategy would implement measures according to their cost-effectiveness so as to reduce the areas of exceedance of these limits. Such a strategy would deliver increasingly smaller areas above the limit values. In the remaining areas, it may well be that reaching complete compliance is very difficult and costly. In addition, there would be little incentive to improve air quality where limit values are already respected.

For pollutants with no effect threshold, such as $PM_{2.5}$, it will generally be more beneficial for public health to reduce pollutant concentrations across the whole of an urban area, as benefits would accrue from reductions in pollution levels even in relatively "clean" areas.

Therefore, an ERT was proposed for fine particulate matter $PM_{2.5}$. $PM_{2.5}$ is responsible for significant negative impacts on human health. Further, there is as yet no identifiable threshold below which $PM_{2.5}$ would not pose a risk. Advice from the WHO suggests that it is justified to assume a linear response linking exposure to $PM_{2.5}$ to adverse effects. This advice should apply both in "clean" and in "polluted" areas. The exposure reduction concept entails a reduction in the exposure of a larger part of the population compared to the limit value approach, which affects (as we approach complete compliance) a smaller number of people. As such, the overall improvement in public health comes at a higher cost with limit values. A Commission Working Group has looked at this issue and concluded that exposure reduction would be a more cost-effective way of reducing air pollution [see chapter 9 of CAFE scenario analysis report no. 4 (Amann et al., 2005)].

However, there is also an issue of environmental justice. Therefore, the European Commission stressed that it is necessary to limit the absolute maximum individual risk for European citizens. This is why the European Commission proposes to keep a limit value in addition to the ERT. The new approach combines:

- A relative target for the reduction of ambient concentrations averaged over a wide geographical area. The extent of this reduction could be determined by the balance of costs and benefits. Intuitively, higher reductions should be required in more polluted areas, without putting disproportional pressure on these areas, and taking into account transboundary aspects. Thus, a percentage reduction would seem appropriate.
- A limit value.

The exposure-reduction approach, including any initiative aimed at improving the accuracy of the exposure-response function, embodies a form of environmental justice, although of a kind different from the ambient air quality standards. As long as there are sources of emission in an urban area, then there will always be differences in exposures due to dilution and dispersion, even if there is uniformity in compliance with ambient standards. If the exposure reduction approach is adopted, and if the reduction amount is required to be the same everywhere, then there will be uniformity in the improvement in exposure, in percentage terms, if not in absolute amounts. In addition, when coupled with a concentration "cap," citizens are guaranteed an absolute minimum standard of air quality to protect them against unduly high risks.

The ERT would provide a better air quality management system than one relying solely on ambient air quality standards. The following benefits (in addition to those already mentioned already) have been identified:

- Source-related emissions reductions would contribute more effectively and not just in areas where there are exceedences of limit values.
- No need to modify the ambient air quality standard as time elapses as the emphasis is on reducing overall exposure, thus saving administrative resources.
- Proposed approach would complement and "fine tune" overall emission ceilings for a Member State or region, which, if implemented, alone would not have the necessary focus on the improvement of public health; i.e., the total emission ceilings might be achieved with a disproportionately small improvement in public health, depending on the spatial relationship between the emission reductions and the populations exposed.

At this stage, no experience with the ERT is available.

*Emission reductions.* Emissions are generally spoken of as a function of the underlying emission generating activity and an emission factor, which depends on the applied technology (including any relevant abatement technology). Emission reductions may aim at the reduction of the activity or may be directed to decrease the specific emissions (often through end-of-pipe technologies).

Due to the uneven distribution of emission sources, pollutants show spatial gradients. These gradients vary also as a function of the atmospheric lifetime of pollutants. There are therefore considerable differences in the scale of relevant sources. Broadly speaking, for pollutants with short atmospheric lifetimes such as ultrafine particles, NO and $NO_2$, local sources may dominate the ambient levels. Longer lived species

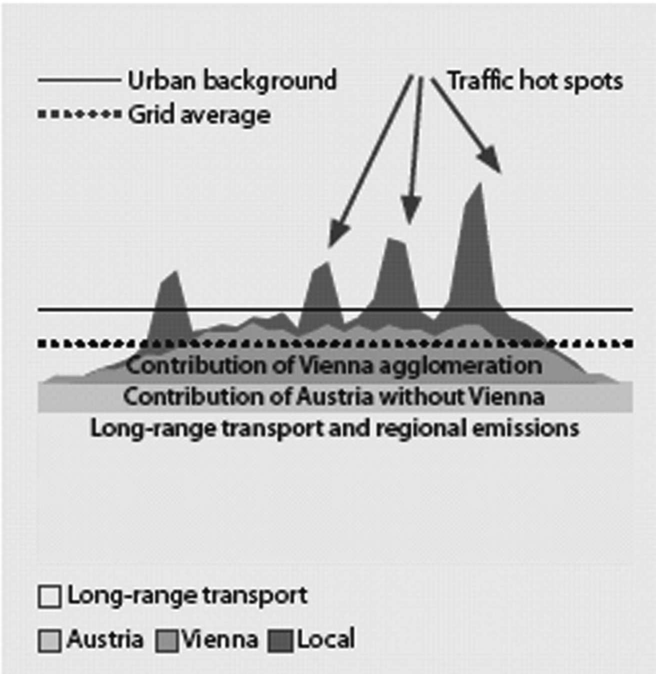



**FIG. 33.** Schematic illustration of different $PM_{10}$ levels in different locations for Vienna (WHO, 2006a).

such as $PM_{2.5}$ and CO may have considerable regional and even continental and hemispheric background levels. This has important implications for control options. The contributions from emissions at different scales are shown schematically in Figure 33 for fine $PM_{10}$. The figure illustrates that reduction strategies at a local scale have only a limited scope.

*Transboundary/Hemispheric Approaches*

It was recognized decades ago that some of the environmental problems linked to air pollution have a strong transboundary component. These problems include acidification (caused by the deposition of oxidized sulfur and nitrogen compounds), eutrophication, and ground-level ozone. Also, fine PM may have a significant transboundary component. This has important consequences for abatement strategies. Since sources and receptors are often located in different countries, multilateral agreements are necessary to combat these effects effectively.

In Europe, the United Nations (UN) ECE Convention on Long Range Transboundary Air Pollution (CLRTAP) provides a framework for emission reduction agreements. Eight protocols have been signed and entered into force in the last decades. Notably, the CLRTAP played also an important role in promoting research to investigate the sources, transboundary transport, and effects of air pollution. The prime objective of these activities was to substantiate cost-effective emission reduction strategies on a European scale. The most recent protocol, the Gothenburg Protocol, set national emission ceilings for four pollutants ($SO_2$, NOx, NMVOC, and $NH_3$) to combat acidification, eutrophication, and ground-level ozone (all these substances are also precursors for secondary PM). While in the past concrete obligations for controlling emissions were derived solely based on technical and economic aspects or equal emission reduction percentages, this protocol attempts to quantify specific reduction requirements for the parties with the aim of achieving certain targets for acidification, eutrophication, and ground-level ozone. The starting points for negotiating emission ceilings were results from a so-called integrated assessment model, which was used to investigate cost-effective emission reductions. The first step was to construct a baseline scenario (including the effect of already decided emissions reduction measures) based on the projected development of emission-generating activities. The emissions are translated into ambient concentrations (using the results of a European dispersion model) and effects in the same modeling framework. In a second step, cost-effective emission reduction strategies (expressed, e.g., as national emission totals in a specific year) can be identified to achieve different environmental improvements. However, the agreed ceilings were in case of the Gothenburg Protocol the result of subsequent political negotiations and did not necessarily reflect cost-effective emission reductions from a European perspective.

In the European Community, the Directive on National Emission Ceilings (NEC) has similar objectives and was based on the same integrated assessment model which was also used for the Gothenburg Protocol. The NEC Directive itself does not contain any concrete requirements for sources. It is up to the Member States to identify those sectors where cost-effective measures should contribute to achieving the ceilings.

*Regional- and National-Level Approaches*

Some measures are usually most effective at a regional or national level. These include many of the source-related regulations listed in Table 6 (such as limit values for installations, national speed limits, etc.). In addition, taxes are usually applied on a national level. This includes fuel taxes or a specific tax on $NO_x$, which, e.g., is applied in Sweden for stationary sources (see later discussion). These instruments have the potential to affect both the emission-generating activities (e.g., mileage of road transport) and the introduction of abatement technologies. Changes in activity usually has ancillary effects (e.g., for road traffic emissions of GHG and noise).

Within the European Community, there is a minimum fuel tax for diesel and petrol. However, the real taxes are often higher and differ by Member States (see Figure 34).

For Austria, the effects of economic instruments for road transport have been assessed and compared to technical measures (such as retrofitting programmes; speed limits; traffic restrictions for high emitting vehicles) to reduce NOx and PM. Notably, a general road pricing scheme for cars and an increase of fuel taxes were among those measures that brought the largest emission reductions.

**TABLE 6**
Categories of measures reported from Member States within plans and programmes

| Sub-category | Measures |
|---|---|
| Category: Traffic | |
| Technical | Emission reduction of cars, buses, trucks, motorcycles, railways, ships, airplanes |
| Traffic management | Traffic flow management, parking charges, congestion charges, improved cargo logistics, airport traffic management |
| Public transport | Improvement and promotion of public transport, promotion of bicycle and pedestrian traffic |
| Traffic restrictions | Measures which restrict traffic in certain areas |
| Road construction | Construction of by-pass roads, constructive measures which improve traffic flow |
| Speed reduction | Area or road specific speed limits |
| Street cleaning | Improved street cleaning, alternative winter sanding |
| Other | alternative traffic concept, bicycle sharing, car sharing, car pooling, efficient driving training, labelling of low emission vehicles, low emission road surface, promotion of methane fuel stations, mobility planning, promotion of railway cargo transport, restrictions to maintain engines running, restrictions to studded tyres, truck toll, tunnel exhaust cleaning |
| Category: Stationary sources | |
| Agriculture | Measures in the area of manure handling and feeding |
| Construction | Measures to reduce emissions on construction sites |
| Heating | Improvement of heaters, building insulation, district heat |
| Industrial | Measures to reduce industrial and power plant emissions |
| Other | Restriction of open fires, removal of sand surfaces |
| Category: Regulation and information | |
| Financial incentives | Fiscal stimulation, emission certificates, financial support of low-emission technology |
| Information of the public | Information and awareness of employees, pupils and the general public |
| Change to emission standards | Improvement of emission standards on the European level |
| Other | — |
| Category: Other measures | |
| Energy | Support of alternative energy production, measures to reduce energy consumption |
| Fuel improvement | Propagation of low-sulphur and low-VOC fuels |
| Urban planning | Integration of mobility and air quality aspects in urban planning. |
| Other | Combination of information, incentives and traffic restrictions; procedure of regularly taking and evaluating new measures. reduction of transboundary pollution; planting of trees; construction of a protective wall. |
| Category: Other activities | |
| Air quality monitoring | Monitoring of pollutant concentrations |
| Studies | Emission inventory, emission monitoring, emission study, energy consumption research, exposure study, research program, study on regional transport |
| Not specified | Measures with unspecified emission reduction, measures which are in the stage of planning |
| Other | Definition of plans to reduce emissions, resettlement of population. |

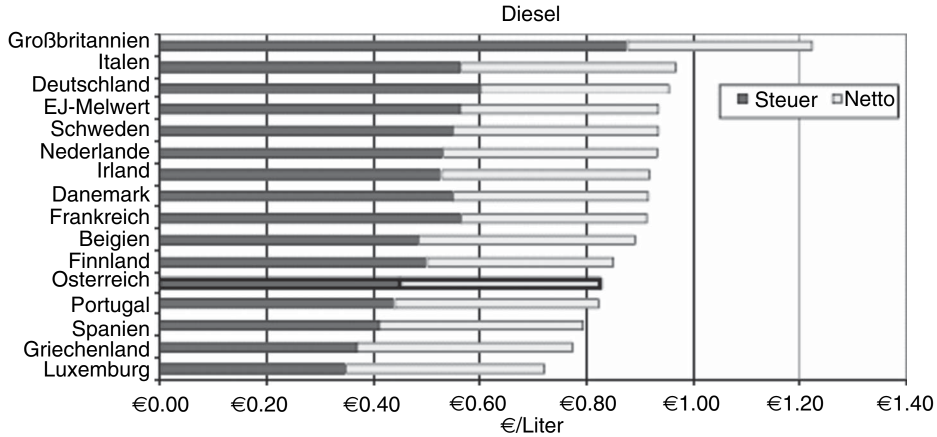



**FIG. 34.** Fuel prices and taxes in different EC Member States; data from autumn 2004.



There are some other successful examples for the application of economic instruments within the European Community. For example, Sweden has implemented a charge on $NO_x$ emissions in order to reduce these emissions cost effectively. According to the $NO_x$ Act the charge is paid for emissions of $NO_x$ from boilers, stationary combustion engines, and gas turbines with a useful energy production with at least 25 GWh per annum. The charge is based on actual recorded emissions and is imposed irrespective of fuel used. It is levied at a rate of SEK 40 (about € 4.3) per kilogram of emitted $NO_x$. To avoid distorting the pattern of competition between those plants that are subject to the $NO_x$ charge and those that are not, the system is designed so that all revenue except the cost of administration is returned to the participating plants, in proportion to their production of useful energy. Boilers with high emissions relative to their energy output are net payers to the system, and sources with low emissions relative to energy output are net recipients. This feature of the system encourages the targeted plants to reduce their emissions of nitrogen oxides per unit of energy to the lowest possible level. Since the Swedish Parliament passed legislation introducing the $NO_x$ charge in June 1990 the specific emissions have dropped from an average of about 160 mg $NO_x$ per megajoule (mg/MJ) of energy input to about 55 mg/MJ, equivalent to 65%.

### *Local Emission Reduction Approaches*

*Point sources and transportation sources.* The most important source-related regulations on the EC level (which have also a considerable impact on local air quality) are the continuous tightened EURO standards for mobile sources and the IPPC and LCP for power generation and industrial installations. However, the EURO standards and the LCP are applicable irrespective of air pollution levels. According to the IPPC Directive, measures going beyond BAT may be requested if this is necessary to achieve EC environmental objectives (such as limit values).

Measures to comply with limit values are usually in addition to these regulations. There are numerous possible additional measures for all relevant sectors. Databases containing lists of possible measures to reduce air pollution at a local scale are now available. These databases often contain estimates for reduction potentials and costs.

The measures reported in plans and programs under the AQ FWD are summarized in Table 6 and Figure 35.

*Management of hot spots.* As stated previously, limit values apply throughout the territory of Member States. Therefore, efforts to comply with limit values are often focused on hot-spot locations (locations in the vicinity of emission sources with the highest pollution levels). As part of the information transmitted by Member States on plans and programs under the Air Quality FWD, the authorities have to quantify the area of exceedance (and for traffic hot spots the length of roads). This information is shown in Figure 36, indicating that some plans aim at the reduction of pollution in rather limited areas.

A recent study on the ex post evaluation of local measures in the EU concluded that:

- Specifically targeted local measures do appear effective in terms of local emissions reductions, air quality improvement, and progress toward legally binding air quality limit values, particularly when these schemes tend to be targeted at air quality hot spots (such as low emission zones; fuel bans; traffic flow controls). They also have good benefit to cost ratios, which are similar to or better than those for the introduction of European level air quality policies. This provides some initial support for these measures as a complement to further European based legislation.
- The effectiveness of all local measures is very site specific. It is not possible to simply transfer schemes between locations without consideration of local conditions. Location-specific characteristics of the following key factors determine this effectiveness:

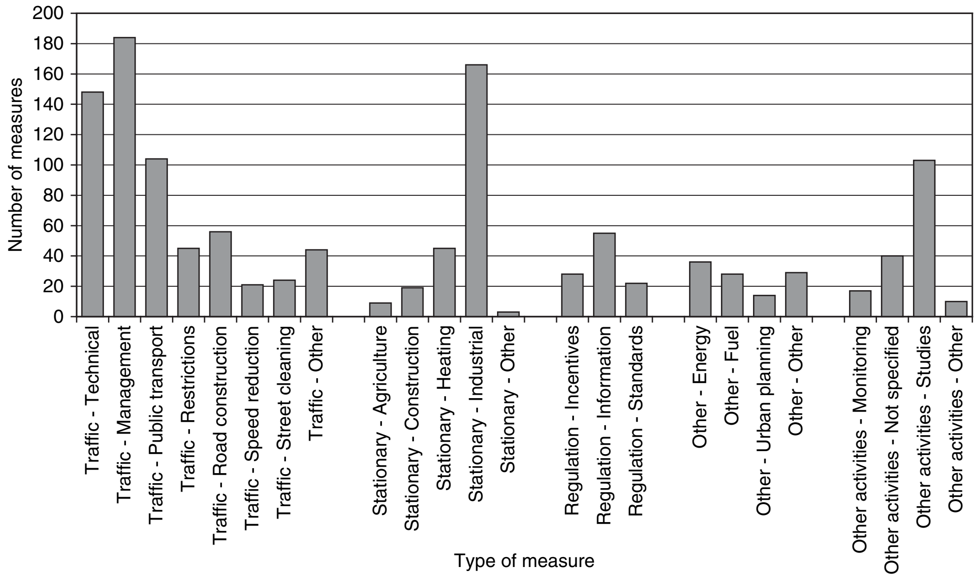



**FIG. 35.** Categories of measures reported by all Member States.

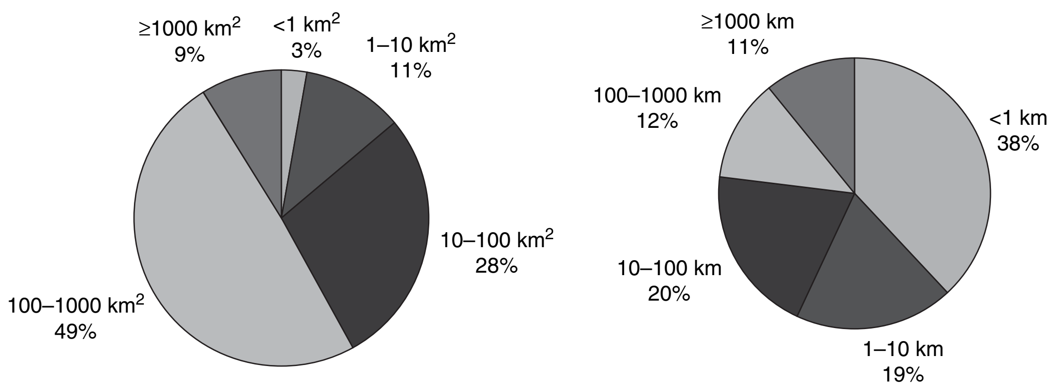



**FIG. 36.** Estimate of the surface area where the level was above the limit value + margin of tolerance in the reference year. Right: Estimate of the length of road where the level was above the limit value + margin of tolerance in the reference year. For 27% of exceedance situations, no surface area and no length of road was reported.



background pollutant levels, pollutant formation and transport mechanisms, and cultural and economic factors influencing the scale and frequency of emissions from various sectors and legal and informational limitations on the ability of responsible authorities to act.

- The schemes most effective in reducing emissions and reducing air quality hot spots appear to be those schemes directly focused on air quality improvements. This includes measures such as low emission zones, motorway flow management, smoky vehicles bans, etc. in urban areas. Many traditional local transport schemes appear less effective in achieving emissions or air quality improvements, though this is not surprising when these schemes are aimed at other problems (e.g., congestion). However, these latter schemes have other benefits (e.g., travel time benefits, reduced accidents, etc.) that are often their primary objective.

*Public Education/Behavioral/Stakeholder Engagement Approaches*

Emissions of air pollutants are often linked to the individual life style of citizens. This includes the choice of the transport mode, and also the use of energy. Many campaigns were launched to influence individual behavior of citizens. It is not easy to find published ex-post reviews of the effectiveness of such campaigns. A small survey among experts in Austria (mainly based on expert judgment) suggests that such campaigns are often limited in their effectiveness. However, there was also a consensus that public communication is an important element in increasing the acceptance of the public for new measures.

## Air Quality Management in Hong Kong

### *Historical Perspective on Air Quality in Hong Kong*

The Hong Kong Special Administrative Region (SAR) is a territory of 1,100 km$^2$ comprising an archipelago of two major islands and many smaller outer islands, a peninsula, and land adjacent to the mainland of the People's Republic of China. Annual deaths total about 30,000, and age-standardized total mortality (0.4%) is about 18% lower than in the West, with cardiovascular disease 47% lower and respiratory disease 40% higher. The annual GDP per capita is US$25,000 in a mostly service-based economy. In recent years, the manufacturing sector has moved north of the boundary into the mainland, and over 70,000 factories operate around the Pearl River Delta.

In 1990, by restricting fuel sulfur content to 0.5% by weight, the Hong Kong SAR demonstrated that even modest reductions in pollution led to significant health gains (Peters et al., 1996; Wong et al., 1998; Wong et al., 1999; Hedley et al., 2002). Since then air quality has been continuously degraded. Despite the progressive establishment of a large evidence base on air pollution health effects, there has been a lack of recognition of the real community costs incurred by harm to health and lost productivity caused by air pollution; a lack of comprehensive approaches to improve urban air quality including cleaner fuels, transportation, and infrastructure of urban environments; and a failure to implement a sufficiently comprehensive range of new laws and regulations on emissions, revise and enforce air quality and standards to update the 1987 Air Quality Objectives, or make significant progress in cross-boundary agreement with the mainland authorities and the business and power sectors in Hong Kong on pollution abatement. A large proportion of heavily polluting factories in the Pearl River Delta region is made up of Hong Kong and foreign business investments. As a result, air quality in Hong Kong now compares unfavorably with the current situation in other world cities such as Auckland, Berlin, London, New York, Paris, and Vancouver. Particulate levels are about 40% higher than in Los Angeles, the most polluted city in the United States (U.S. EPA, 2003).

## Visibility, Air Pollutants, and Health

Watson (2002) cites the U.S. Environmental Protection Agency (EPA) as identifying impaired visibility as the "best understood of all environmental effects of air pollution." In addition to impairing quality of life, daily loss of visibility directly reflects the risk of injury by airborne pollutants on cardiovascular and pulmonary systems. The commonest manifestations of these health problems include serious cardiopulmonary events such as heart attacks, stroke, and respiratory illnesses, including bronchitic symptoms of cough, phlegm, and wheeze, acute and chronic bronchitis, pneumonia, and attacks of asthma.

Effects of air pollution on visibility are apparent to everyone, but the health effects may be silent and unobservable until they result in symptoms, illness episodes, and death. Even then, direct attribution of illness in an individual with daily air pollution is not possible in the same way as it would be with infectious disease. This uncertainty and lack of direct evidence are associated with lower perceptions of risk by some sections of the public who are potentially important drivers of policy and with lack of political will by decision makers, and it provides scope for arguments against interventions and air quality controls by vested interests.

Visibility has been deteriorating in Hong Kong for several years (Figure 37). This trend is now raising concern in the tour-

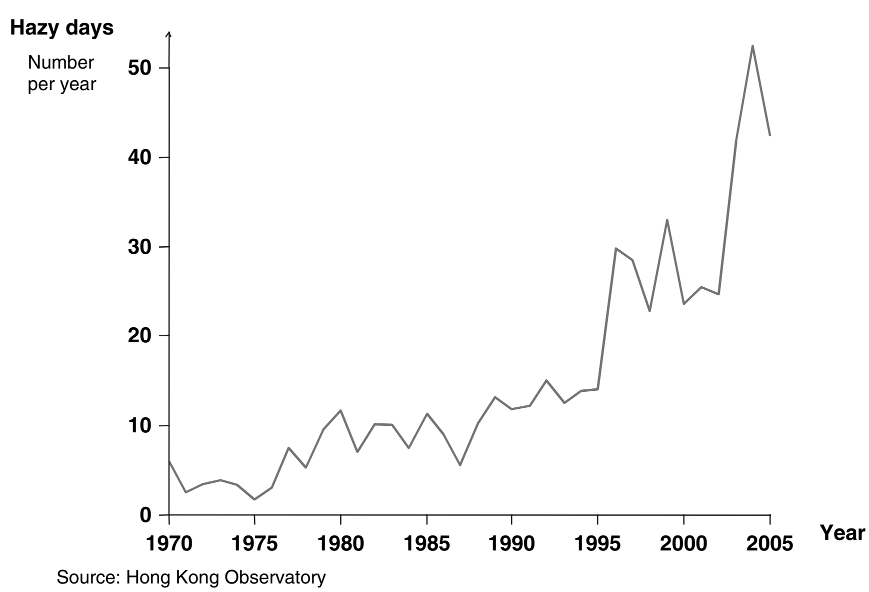



**FIG. 37.** Deteriorating visibility (<8km) due to haze based on direct observation of landmarks by the Hong Kong Observatory 1970–2005.

ism and hospitality industry because of its impact on Hong Kong's attractiveness as a destination, and in the finance and foreign business sector because of increasing difficulties in recruiting overseas personnel. However, the threat of pollution to the health of the local community, demonstrated in many recent scientific reports, has not prompted the necessary radical action.

*Case Study: Visibility as a Tool for Air Quality Management in Hong Kong*

For the source of this description, see Hedley et al. (personal communication).

The Hong Kong SAR and the mainland air quality objectives (AQO) are long outdated and provide no health protection from pollution. The Hong Kong AQO (Hong Kong Environmental Protection Department) date from 1987 and were based on the standards adopted by the U.S. Environmental Protection Agency. None of the subsequent international revisions of air quality standards have been reflected in any changes to the HKSAR objectives. In addition they have come to be regarded as legal, safe, and permissive levels in environmental impact assessments. Through that mechanism the AQO have become an instrument by which air pollution in different zones may actually be legally increased under the Environmental Impact Assessment Ordinance.

For some time, the local community in Hong Kong has raised concerns about levels of air pollution, but the use of outdated standards as the basis for a daily air pollution index (U.S. EPA, 2006b) has prevented communication of the true degree of hazard associated with current pollution levels. By using an easily understood and highly apparent indicator such as visibility and associating different levels of visibility with potential changes in costs incurred, our aim is to promote a better understanding of the impacts of air pollution.

As part of a program of public accountability we used photographs on poor and better visibility days as representations of the relationships between visibility, air pollution, health effects, and community costs for health care and lost productivity (Hedley et al., 2008). We used coefficients from time-series models and gazetted costs to estimate the health and economic impacts of different levels of pollution. In this population of 6.9 million, air quality improvement from the annual average to the lowest pollutant levels of better visibility days, comparable to the World Health Organization air quality guidelines, would avoid 1335 deaths, 60,587 hospital bed days, and 6.7 million doctor visits for respiratory complaints each year. Direct costs and productivity losses avoided would be over US$246 million a year and US$2250 million for intangible costs. The dissemination of these findings led to increased demands for pollution controls from the public and legislators but denials of the need for urgent action from government, which has implemented its own review of the health effects of air pollution health effects that will take more than 2 yr to complete. The outcome demonstrates the need for more effective translation of the scientific evidence base into risk communication and public policy.

## Evidence of Effectiveness of Air Quality Management Interventions

*North America: Steel Mill Closure in Utah Valley*

During the period of August 1986 to Sept 1987, a steel mill, which was the primary source of particulate pollution in Utah Valley, was closed for a 13-month period due to a workers' strike. The effects of the closure and subsequent reopening of this mill on air quality and hospital admissions for respiratory diseases among children were investigated (Pope, 1989). The study period covered 12 mo before and 12 mo after the closure period. This study focused on the results from the winter seasons, since $PM_{10}$ levels are typically highest during the winter. The two winter seasons before closure and after reopening experienced 13 and 10 exceedances, respectively, of the U.S. federal 24-h $PM_{10}$ standard (150 $\mu g/m^3$). In contrast, during the closure period, the standard was not exceeded. When hospital admissions were analyzed for the same time periods, striking differences were observed.

During the 1986/1987 winter season, when the mill was closed, hospital admissions for children were approximately 3 times lower than when it was open. Statistical analyses showed that this decrease was associated with the decrease in $PM_{10}$ levels.

*North America: Traffic Reduction in Atlanta, GA, During Olympic Games*

During the 17 days of the summer Olympic Games in Atlanta, GA, traffic patterns changed due to the alternative transportation strategy that was implemented to relieve traffic congestion. Researchers analyzed the effects of these changes on air quality and acute asthma events among children by examining the air quality and hospital records 4 wk before and 4 wk after the Games (Friedman et al., 2001).

Ambient ozone concentrations measured at 3 monitoring sites decreased by approximately 13% during the Olympic Games. Carbon monoxide (1.26 vs 1.54 ppm, 19% decrease, $p$ = .02) and $PM_{10}$ (30.8 vs 36.7 $ug/m^3$, 16% decrease, $p$ = .01) concentrations also declined significantly, while the decline in $NO_2$ levels was not significant (36.5 vs 39.2 ppb, 7% decrease, $p$ = .49) and $SO_2$ levels increased (4.29 vs 3.52, 22% increase, $p$ = .65).

Correspondingly, there were fewer children admitted to the hospitals for acute asthma, an average of 2.5 cases per day during the Olympic Games compared to 4.2 cases per day in the baseline period (before and after the Olympic Games). The study determined that there were no significant changes in weather conditions or emissions from stationary sources. Also, hospital admissions for other causes among children did not change during this period.

*North American: California Children's Health Study*

The Children's Health Study, which began in 1992, is a large, long-term study of the effects of chronic air pollution exposures on the health of children living in southern California. In one of the studies, investigators examined the health effects of relocating to areas of differing levels of air pollution (Avol et al., 2001). They followed 110 children from the larger Children's Health Study who moved to 6 Western states at least 1 yr before follow-up and to areas of either higher or lower pollution. They found that children moving to areas with lower $PM_{10}$ levels experienced an increase in lung function growth rates. Conversely, moving to areas of higher $PM_{10}$ resulted in a decrease in lung function growth rate. The results support the view that changes in ambient pollution levels (in this case, $PM_{10}$) may have measurable effects on longer-term lung function (and health) outcomes.

*Europe: United Kingdom*

A UK consultant investigated the effects of short-term and local measures to reduce air pollution (AEAT Environment, 2005). The study concluded that it is extremely difficult to find reliable and consistent data on the ex post costs and the ex post benefits (particularly in relation to emissions and air quality) of local measures.

Probably it is even more difficult to assess and compare consistently the ex post costs and the ex post benefits for local, regional, and national measures. However, a few conclusions can be drawn:

- It is generally accepted that air quality management has been a success story in the EC. Member States of the European Union spend large sums for air quality protection, mainly triggered by source related legislation.
- The EURO standards are seen as an essential element of AQ protection that ensured that the continuously increasing road transport is emitting less pollution than a few years ago even though the EURO standards proved to be less efficient in practice than expected. However, there is still no legislation in force forcing external costs of road traffic to be internalized.
- The concept of integrated pollution prevention for industrial installations including the application of Best Available Technology (BAT) for new and existing plants is also widely accepted. However, there is still a debate if legally binding emission limit values are warranted for those installations.
- Air quality limit values are one important element of air quality policy. The inherent focus on the most polluted sites has been discussed recently and led to the proposal to supplement the limit value approach by an exposure reduction target (ERT), which sets objectives for relative improvements (more or less irrespective of absolute pollution levels) for urban background locations.
- Energy efficiency will become increasingly important (primarily due to concerns about energy prices, security of supply, and climate change).

There is robust evidence indicating that air pollution still causes severe health and environmental damage. Since many measures to reduce air pollution are already in force, this leads to a situation where additional measures are getting increasingly expensive, while the reduction potentials get smaller and smaller. As a consequence, additional measures need to be well justified. This implies that any additional measures need to be based on robust science. This includes a profound knowledge of:

- The sources of air pollution.
- The atmospheric dispersion.
- Ambient levels.
- Effects of air pollutants.
- Costs and reduction potential for abatement measures.

Therefore, recent legislative proposals in the EC have been accompanied by impact assessments comparing the cost and benefits of these proposals.

*Europe: Coal Ban in Irish Cities*

On September 1, 1990, the Irish government banned the marketing, sales, and distribution of soft coal within the city of Dublin. Clancy et al. (2002) examined the effect of this intervention on the association between ambient air quality and death rates. The ban on coal sales resulted in a substantial reduction in black smoke, which is a measure of fine particles. Overall, the average black smoke level fell by about two-thirds after the ban. Similarly, sulfur dioxide levels decreased by about one-third after the ban. The investigators analyzed data from 6 yr prior to and 6 yr after the ban. After adjusting for factors known to influence mortality, which include temperature, relative humidity, respiratory epidemics, age, and changes in personal habits such as smoking, the investigators found statistically significant decreases in death rates. They found a 6% decrease in nontrauma deaths. This decrease was primarily driven by an estimated 10% and 16% decrease in the rates of death from heart and lung diseases, respectively. This finding is consistent with our understanding of air pollution effects on the cardiovascular and respiratory systems. Moreover, the reduction in death rates was two to three times greater than had been predicted from previous PM mortality studies. These findings suggest that control of particulate air pollution can lead to immediate and significant reductions in death rates.

*Asia: Cleaner Fuel in Hong Kong*

On July 1, 1990, all power plants and road vehicles in Hong Kong were restricted to use of fuel oil with a sulfur content of not more than 0.5% by weight. This intervention led to an immediate improvement in air quality, as sulfur dioxide concentrations measured at multiple sites fell an average of 53% over the following year compared to the

baseline levels measured 2 yr prior (Hedley et al., 2002). Concentrations of vanadium and nickel also declined sharply (Hedley et al., 2004). In the 2 yr following the intervention, a reduction of chronic bronchitic symptoms in children and adults (Peters et al., 1996; Wong et al., 1999) and improved lung function in primary school children (Wong et al., 1998) were shown. The impact of the regulation on mortality was assessed by examining death rates between two age groups for the period 1985 through 1995—which includes a 5-yr period before and 5-yr period after the restriction of sulfur content. In the 12 mo following the restriction, seasonal deaths were substantially reduced, followed by a peak in cool-season death rate between 13 and 24 mo, returning to the expected pattern during yr 3–5. Compared with predictions, the intervention led to a significant decline in the average annual trend in deaths from all causes (2.1%; $p = .001$), respiratory (3.9%; $p = .0014$), and cardiovascular (2.0%; $p = .0214$) diseases, but not from other causes. It was estimated that the regulations had resulted in a gain in the average life expectancy of 20 and 41 d in women and men, respectively, for every year of exposure to the lower levels of pollution (Hedley et al., 2002).

The Hong Kong intervention provides direct evidence that even modest reductions in emissions of sulfur dioxde and metals following restrictions on sulfur rich fuels are associated with significant immediate and long-term health benefits.

## Conclusions

This section described how air pollution problems are managed within North America, the European Community, and Asia by presenting both general policy approaches for each continent and detailed case studies for large urban centres. While each area has a unique set of problems—and approaches and capacities to deal with them—there is a clear portfolio of comprehensive management strategies common to successful programs. These include the establishment of ambient air quality standards that define clean air goals, strong public support leading to the political will to address these problems, technology-based and technology-forcing emission limits for all major contributing sources, and enforcement programs to ensure that the emission standards are met.

Initially, many regions focused their air pollution control efforts on lead, ozone, and large particles (i.e., TSP, $PM_{10}$). However, newer epidemiological studies of premature death, primarily conducted in the United States with cohorts as large as half a million participants, have made it clear that long-term exposure to $PM_{2.5}$ is the major health risk from airborne pollutants. While WHO, U.S. EPA, Environment Canada, and California Air Resources Board (CARB) rely on the same human health effects literature, there are striking differences, up to a factor of three, in the ambient air quality standards they set. In addition, how these standards are implemented (e.g., allowable exceedances, natural and exceptional event exceptions) can greatly reduce their stringency. Now there is increasing evidence that there is no level below which exposure to some pollutants has no potential health effects. This will have implications for how some pollutants are regulated. Despite these issues, the ambient air quality standards and the regulatory authorities that result from public and political support have been the major driver of clean air progress.

Worldwide, command and control has been the primary regulatory mechanism to achieve emission reductions, although the European Community has successfully used tax incentives and voluntary agreements with industry. Over the past four decades, the California Air Resources Board set the bar for U.S. EPA and European Union motor vehicle emission standards that are now being adopted in many developing countries, particularly in Asia. Emissions of VOC and CO (and to a lesser extent $NO_x$) from new passenger vehicles were reduced by a factor of a hundred in comparison to precontrol vehicles. The United States adopted emission standards for 2007 and subsequent model year heavy-duty engines that represent 90% reductions of $NO_x$ and PM compared to 2004 model year emission standards. Implementation of reformulated gasoline and diesel fuels resulted in further reductions. Stationary source $NO_x$ and $SO_x$ emission standards were reduced by at least a factor of 10 since 1980. Small off-road engines, architectural coatings, consumer products, and solvents are also targeted for large emission reductions.

Since the emission standards are technology-based or technology-forcing, industry has been able to pursue the most cost-effective strategy to meeting the emission target. As a result, actual control costs are generally less than originally estimated. Over the past three decades, California's motor vehicle and fuel regulations have had a fairly uniform cost over time. In the United States, total air pollution control costs are about 0.1% of GDP, although this has not necessarily resulted in overall job and income loss because the air pollution control industry is about the same size. In addition, the U.S. EPA estimated that each dollar currently spent on air pollution control results in about $4 of reduced medical costs, as well as the value assigned to avoided premature deaths. In the past (1970–1990), when lead reductions and other major control programs were implemented, the benefit to cost ratio was $90 to $1.

An alternative to command-and-control regulations is market-based mechanisms that results in more efficient allocation of resources. The $SO_2$ cap and trade program in the United States resulted in rapid emissions reduction at lower cost than was initially anticipated. Efforts to extend the cap and trade system to $SO_2$, mercury, and $NO_x$ emissions in the eastern United States were less successful due to several issues related to heterogeneous emissions patterns that could worsen existing hot spots, allocation of emissions allowances, procedures for setting and revising the emissions cap, emissions increases following transition to a trading program, and compliance assurance.

Emission reduction initiatives at the local level also play a critical role in air quality management. Local governments can contribute to cleaner air through emission reduction measures aimed at corporate fleets, energy conservation and efficiency measures in

municipal buildings, public education to promote awareness and behavior change, transportation and land use planning, and bylaws (anti-idling etc.). Many large urban centers such as the City of Toronto are following the policy trend toward an integrated and harmonized approach to cleaner air and lower greenhouse gas emissions.

A comprehensive enforcement program with mandatory reporting of emissions, sufficient resources for inspectors and equipment, and meaningful penalties for noncompliance ensures that emission standards are being met. While air quality management through standards for vehicles and fuels have resulted in measurable reductions in emissions, regulation of emissions for in-use vehicles through I/M programs poses greater technical challenges.

An evidence-based public health approach in the assessment of health impacts of air pollution may not lead to essential policy changes. Environmental advocacy must develop more effective methods of risk communication to influence public demand for cleaner air and strengthen political will among decision makers.

Average daily visibility has been declining in Asia over two decades. Visibility provides a measure, with face validity, of environmental degradation and impaired quality of life, and a risk communication tool for pollution-induced health problems, lost productivity, avoidable mortality, and their collective costs.

Although scarce, information from both planned and unintended air quality interventions provides strong evidence in support of temporal association and causality between pollution exposures and adverse health outcomes. Even modest interventions, such as reductions in fuel contaminants and short-term restrictions on traffic flows, are associated with marked reductions in emissions, ambient concentrations, and health effects. Coal sales bans in Ireland and fuel sulfur restrictions in Hong Kong, successfully introduced in large urban areas within a 24-h period, were economically and administratively feasible and acceptable, and effective in reducing cardiopulmonary mortality.

In response to severe air quality problems, many urban centers imposed comprehensive emission controls, but growth in population, energy demand, vehicle miles traveled, industrial activity, and aging vehicle fleets prevented attainment of all the health-based ambient air quality standards. While some air quality problems have been eliminated or greatly reduced (i.e., lead, $NO_2$, $SO_2$), particulate matter and ozone levels remain high in many large cities, resulting in hundreds of thousands of deaths per year and increased disease rates. In response, air quality management agencies are developing innovative approaches, including regulation of in-use emissions, reactivity-based VOC controls, and exposure-based prioritization of PM controls. Several cooperative, multinational efforts have begun to address transboundary issues. Newly recognized challenges also need to be integrated into air quality management programs, ranging from the microscale (e.g., air pollution "hot spots," ultrafine particles, indoor air quality) to global scales (e.g., climate change mitigation, international goods movement).

## Key Messages

- While North America, the European Community, and Asia have a unique set of air pollution problems—and approaches and capacities to deal with them—there is a clear portfolio of comprehensive management strategies common to successful programs. These include the establishment of ambient air quality standards that define clean air goals, strong public support leading to the political will to address these problems, technology-based and technology-forcing emission limits for all major contributing sources, and enforcement programs to ensure that the emission standards are met.
- Initially, many regions focused their air pollution control efforts on lead, ozone, and large particles (i.e., TSP, $PM_{10}$). However, newer epidemiological studies of premature death, primarily conducted in the United States with cohorts as large as half a million participants, have made it clear that long-term exposure to $PM_{2.5}$ is the major health risk from airborne pollutants. While WHO, U.S. EPA, Environment Canada, and California Air Resources Board (CARB) rely on the same human health effects literature, there are striking differences, up to a factor of three, in the ambient air quality standards they set. In addition, how these standards are implemented (e.g., allowable exceedances, natural and exceptional event exceptions) can greatly reduce their stringency.
- Worldwide, command and control has been the primary regulatory mechanism to achieve emission reductions, although the European Community has successfully used tax incentives and voluntary agreements with industry. Over the past four decades, the California Air Resources Board set the bar for U.S. EPA and European Union motor vehicle emission standards that are now being adopted in many developing countries, particularly in Asia.
- Since the emission standards are technology-based or technology-forcing, industry has been able to pursue the most cost-effective strategy to meeting the emission target. As a result, actual control costs are generally less than originally estimated. In the United States, total air pollution control costs are about 0.1% of GDP, although this has not necessarily resulted in overall job and income loss because the air pollution control industry is about the same size. In addition, the U.S. EPA estimated that each dollar currently spent on air pollution control results in about $4 of reduced medical costs as well as the value assigned to avoided premature deaths
- A comprehensive enforcement program with mandatory reporting of emissions, sufficient resources for inspectors and equipment, and meaningful penalties for noncompliance ensures that emission standards

are being met. While air quality management through standards for vehicles and fuels have resulted in measurable reductions in emissions, regulation of emissions for in-use vehicles through inspecton and maintenance (I/M) programs poses greater technical challenges.

- An alternative to command-and-control regulations is market-based mechanisms that result in more efficient allocation of resources. The $SO_2$ cap and trade program in the United States resulted in rapid emissions reduction at lower cost than was initially anticipated. Efforts to extend the cap and trade system to $SO_2$, mercury and $NO_x$ emissions in the eastern United States were less successful due to several issues related to heterogeneous emissions patterns that could worsen existing hot spots, allocation of emissions allowances, procedures for setting and revising the emissions cap, emissions increases following transition to a trading program, and compliance assurance.
- Emission reduction initiatives at the local level also play a critical role in air quality management. Local governments can contribute to cleaner air through emission reduction measures aimed at corporate fleets, energy conservation and efficiency measures in municipal buildings, public education to promote awareness and behavior change, transportation and land use planning, and bylaws (anti-idling etc). Many large urban centers such as the City of Toronto are following the policy trend toward an integrated and harmonized approach to cleaner air and lower greenhouse gas emissions.
- An evidence-based public health approach in the assessment of health impacts of air pollution may not lead to essential policy changes. Environmental advocacy must develop more effective methods of risk communication to influence public demand for cleaner air and strengthen political will among decision makers.
- Average daily visibility has been declining in Asia over two decades. Visibility provides a measure, with face validity, of environmental degradation and impaired quality of life, and a risk communication tool for pollution-induced health problems, lost productivity, avoidable mortality, and their collective costs.
- Although scarce, information from both planned and unintended air quality interventions provides strong evidence in support of temporal association and causality between pollution exposures and adverse health outcomes. Even modest interventions, such as reductions in fuel contaminants and short-term restrictions on traffic flows, are associated with marked reductions in emissions, ambient concentrations and health effects. Coal sales bans in Ireland and fuel sulfur restrictions in Hong Kong, successfully introduced in large urban areas within a 24-h period, were economically and administratively feasible and acceptable, and effective in reducing cardiopulmonary mortality.
- While some air quality problems have been eliminated or greatly reduced (i.e., lead, $NO_2$, $SO_2$), particulate matter and ozone levels remain high in many large cities, resulting in hundreds of thousands of deaths per year and increased disease rates. Air quality management agencies are developing innovative approaches, including regulation of in-use emissions, reactivity-based VOC controls, and exposure-based prioritization of PM controls. Several cooperative, multinational efforts have begun to address transboundary issues. Newly recognized challenges also need to be integrated into air quality management programs, ranging from the microscale (e.g., air pollution "hot spots," ultrafine particles, indoor air quality) to global scales (e.g., climate change mitigation, international goods movement).

## EMERGING CHALLENGES AND OPPORTUNITIES IN THE DEVELOPMENT OF CLEAN AIR POLICY STRATEGIES

### Introduction

There are a number of challenges and opportunities facing decision makers in the development of clean air policy strategies, particularly when dealing with specific pollutants, the linkages between air issues, and how best to address them at various spatial scales. To date most of the attention in the literature and among policymakers has been on addressing specific pollutants and air issues (e.g. the precursors to ground level ozone, acid rain), and in some cases even adopting a multipollutant approach as in the UNECE Convention on Long Range Transboundary Air Pollution. These issues tend to operate at the local, regional, or airshed scale, and in some cases they may cross international borders, thereby requiring a binational or multinational response.

In recent years, however, the issue of air quality management is beginning to take on global dimensions, as scientific evidence mounts regarding the wide dispersion and deposition of hazardous airborne pollutants (HAPs), air toxics, and persistent organic pollutants (POPs). An emerging issue that is receiving multinational attention is hemispheric air pollution transport, where air pollutants are transported across oceans and contribute to local/regional air quality problems in jurisdictions on another continent, thousands of kilometers away. An even greater concern, however, is the global threat of climate change, which has the potential to be the most significant environmental issue facing humankind. While climate change has direct implications for air quality, air pollutants can also greatly impact climate change, via the greenhouse gases (GHGs) and some aerosols such as black carbon causing warming effects, and in other cases via sulfate aerosols, which

have a significant regional cooling effect. Not surprisingly, the linkages between climate change and air pollution, how to control their source pollutants (GHGs and criteria air contaminants), and how they may interact to pose a cumulative risk to human health are emerging as important challenges to air quality management.

This section outlines many of the challenges for air quality management on local urban scales, and also extends the discussion to wider spatial scales, while considering the important linkages between air quality and climate change policies. The focus is on urban air quality management, with specific reference to particulate matter. The section discusses novel approaches to air quality management, including the issue of environmental justice and the policy challenges arising from the management of hemispheric air pollution transport; it also discusses the linkages between air quality and climate change, including the opportunities that coupling these air issues provides for air quality management. The section concludes with a brief discussion on future research requirements. Although examples are largely drawn from the EU (especially the United Kingdom) and North America (especially Canada), the challenges and opportunities apply to both developed and developing countries.

### Urban Air Quality Management

In many areas of the developed world air quality management is a fairly mature subject. There have been some recent developments that are of significance to these areas, as well as to developing countries where air quality management systems may be at an earlier stage. A particularly notable development in this context has been the publication of a global update of the World Health Organization guidelines for air quality (WHO, 2006b). The significant feature of this publication in the current context is that it explicitly addresses the problems of air quality management in developing countries. An example of this is the way the WHO has dealt with particulate matter (PM). In its previous publication of air quality guidelines in 2000, the WHO recommended guidance for risk management in the form of exposure-response relationships and suggested that air quality managers quantify the risks relevant to local levels of PM and make the risk management decisions on control policies appropriate to whatever balance of risks and benefits was felt appropriate. While this approach is used in some of the developed countries and regions, in the 2006 update there was a body of opinion presented to the WHO, largely from the developing countries, which felt that this level of detail was not particularly helpful. Accordingly, the 2006 update has now returned to the older approach of recommending a guideline expressed as a concentration and averaging time, together with a series of three successively more stringent "interim targets" that approach the guideline. WHO recognizes this approach as particularly helpful for developing countries whose levels of PM are currently quite some way above the guidelines themselves.

There are several significant implications of this approach for air quality and risk managers. First, the move away from recommending an exposure-response coefficient or function for a nonthreshold pollutant such as PM could be viewed in some quarters as not allowing any scope for national or regional air quality managers to undertake their own risk management and to formulate their policy targets considering local prevailing levels of pollution as well as the predominant socioeconomic climate. This latter approach is the way policies for the management of PM levels are handled in the United Kingdom. For example, expert advisory groups under the umbrella of the UK Department of Health have devoted considerable intellectual resource to assessing the literature and recommending exposure-response coefficients, together with likely uncertainty ranges, so that in formulating policy measures, the Environment Department can undertake the appropriate quantification of the effects on public health and, where possible, cost-benefit analysis and proceed with policies that respect the three "pillars" of sustainable development. This process can still be undertaken, of course, but there is now no international body recommending an exposure-response relationship to underpin such risk management analyses. (It is worth noting, however, that the UNECE/WHO Task Force on Health within the Convention on Long Range Transboundary Air Pollution did assess the then current literature and recommended exposure-response relationships in 2004.)

To carry out risk management analyses of the type just described requires a fairly well developed scientific and economic infrastructure in the air quality management system. The assessment of potential new control policies proceeds according to the following impact pathway chain:

Possible policy scenario → projected emissions
→ atmospheric concentrations/ exposures
→ health and/or environmental effects
→ monetized damage costs vs. control costs

The first stage requires estimates of the effect on emissions of the proposed policies, the second requires, at the very least, a robust dispersion/chemical model of atmospheric transport, the third requires one or more exposure-response functions, and the final stage requires economic analysis of the possible monetary values associated with the effects of the policies. This chain therefore requires a certain maturity of development in the air quality management system. The discussions in the WHO forum that produced the global update indicated that participants felt that this was not appropriate to less well developed systems in many countries and therefore chose to recommend the Guideline and Interim Targets for PM referred to earlier. The possible criticism that these concentration levels were somewhat arbitrary (in the sense that they were not

derived from a balance of costs and benefits as might be thought appropriate for a nonthreshold pollutant) was addressed by choosing the guideline for $PM_{2.5}$ as the lowest level at which total, cardiopulmonary, and lung cancer mortality were shown to increase with more than 95% confidence in the ACS study of Pope et al. (1995) The highest value (i.e., least stringent) interim target, denoted IT-1, was chosen as the highest observed value in the studies on long-term health effects. The two values in between, IT-2 and IT-3, were essentially arbitrary values chosen between the other two extremes; the change in risk on moving from one level to the next was quoted, using essentially the exposure-response relationships derived by Pope et al. in the ACS study.

While in the short term this approach will no doubt be helpful to developing countries in formulating policies, in the longer term it may well need revision as the policy process matures and both sides of the debate become more knowledgeable and sophisticated. There will be further challenges for developed countries in future too, as they approach the lower end of the IT/guideline scale produced by WHO. The potential difficulty of deriving conclusions of policy relevance from epidemiological studies as levels of PM decrease has been noted in the second section and this is likely to complicate the policy process. For example, let us suppose a country or region approaches the guideline for $PM_{2.5}$ (10 $\mu g/m^3$ annual mean). Does it stop once the level is reached or will there still be robust epidemiological evidence for the absence of a threshold that suggest further improvements might be warranted? If so, what does one make of the guideline? Questions of this kind will make the management of PM a subject of some difficulty for years to come. As levels continue to decrease, the wider issues of where society should best deploy scarce resources in improving public health and environmental protection will become increasingly important (see, for example, Krupnick, 2008).

## Novel Approaches to Air Quality Management

The final paragraph above serves as a useful introduction to the topic discussed in this section, which was also prefaced in the second section. This concerns the problem of how one manages air quality in areas where standards or guidelines are already met, or where there remain some areas where complete compliance is extremely difficult or expensive. This is particularly important for nonthreshold pollutants like PM, and highlights the problems associated with managing air quality via a single air quality standard or guideline. This problem becomes increasingly apparent as ambient levels are successively reduced and the easier control measures have been taken.

Experience in the United Kingdom and elsewhere in Europe has shown that for nonthreshold pollutants, single limit values or standards may not on their own be the most appropriate way of managing air quality, particularly in areas where existing air quality management systems are mature. This has encouraged the European Commission to propose a new, additional concept, the exposure reduction target (ERT) which was adopted in April 2008. The following short description of the basics of the concept is derived from a non-paper issued by the European Commission.

The existing legal framework of the Air Quality Framework Directive and its daughter directives (*Official Journal of the EC*, 1996 and 1999, respectively) require complete compliance, meaning that limit values must be met everywhere. As such, a conventional air quality management strategy would implement measures according to their cost-effectiveness so as to reduce the areas of exceedence of these limits. Such a strategy would deliver increasingly smaller areas above the limit values. In the remaining areas, it may well be that reaching complete compliance is very difficult and costly. In addition, there would be no legal requirement to improve air quality where limit values are already respected.

For pollutants with no effect threshold, such as $PM_{2.5}$, it will generally be more beneficial for public health to reduce pollutant concentrations across the whole of an urban area as benefits would accrue from reductions in pollution levels even in relatively "clean" areas. Therefore, an ERT was proposed by the European Commission for fine particulate matter $PM_{2.5}$. $PM_{2.5}$ is responsible for significant negative impacts on human health. Further, there is as yet no widely agreed threshold below which $PM_{2.5}$ would not pose a risk. Advice from the WHO suggests that it is justified to assume a linear response linking exposure to $PM_{2.5}$ to adverse effects. This advice should apply both in "clean" as well as in "polluted" areas. The exposure reduction concept entails a reduction in the exposure of a larger part of the population compared to the limit value approach, which affects (as we approach complete compliance) a smaller number of people. As such, the overall improvement in public health comes at a higher cost with limit values. A Commission Working Group has looked at this issue and concluded that exposure reduction would be a more cost-effective way of reducing air pollution [see chapter 9 of CAFE scenario analysis report no. 4 (Amann et al., 2005)].

## Environmental Justice

There is also an issue of environmental justice. The European Commission has stressed that it is necessary to limit the absolute maximum individual risk for European citizens. This is why the European Commission proposes to keep a limit value, in addition to the ERT. The new approach combines:

- A relative target for the reduction of ambient concentrations averaged over a wide geographical area. The extent of this reduction could be determined by the balance of costs and benefits. Intuitively, higher reductions should be required in more polluted areas, without putting disproportional pressure on these areas, and taking into account transboundary aspects. Thus, a percentage reduction would seem appropriate.
- A limit value.

In order that the new system does not return in practice to the old inefficient standards-based system, the limit value or "cap" ought not to be the single major driver of air quality management policies. The exposure-reduction approach, including any initiative aimed at improving the accuracy of the exposure-response function, embodies a form of environmental justice, although of a different kind from the ambient air quality standards. As long as there are discrete sources of emission in an urban area, then there will always be differences in exposures due to dilution and dispersion, even if there is uniformity in compliance with ambient standards. If the exposure reduction approach is adopted, and if the reduction amount is required to be the same everywhere, then there will be uniformity in the improvement in exposure, in percentage terms, if not in absolute amounts. In addition, when coupled with a concentration "cap," citizens are guaranteed an absolute minimum standard of air quality to protect them against unduly high risks.

The ERT would provide a better air quality management system than one relying solely on ambient air quality standards. The following benefits (in addition to those already mentioned) have been identified:

- Source-related emissions reductions would contribute more effectively and not just in areas where there are exceedences of limit values.
- No need to modify the ambient air quality standard as time elapses, as the emphasis is on reducing overall exposure, thus saving administrative resources.
- The proposed approach would complement and "fine tune" overall emission ceilings for a Member State or region, ceilings that if implemented alone would not have the necessary focus on the improvement of public health; i.e., the total emission ceilings might be achieved with a disproportionately small improvement in public health, depending on the spatial relationship between the emission reductions and the populations exposed.

## Hemispheric Air Pollution Transport and Policy Challenges

In the last few years there has been a growing recognition that transport of air pollutants (as opposed to long-lived greenhouse gases) can occur between continents, particularly in the Northern Hemisphere. This presents a challenge to the scientific community but also to the policymaker. One step forward in understanding this problem, initially from a scientific point of view, was the establishment in December 2004 of the Task Force on Hemispheric Air Pollution within the Convention on Long Range Transboundary Air Pollution (CLRTAP) of the United Nations Economic Commission for Europe. The discussion in this section is based largely on a paper produced by the cochairs of the Task Force, Terry Keating (U.S. EPA) and Andre Zuber (European Commission) (Zuber & Keating, 2005).

There is well-documented evidence for the intercontinental transport of ozone, particles, and their precursors, as well as mercury and some persistent organic pollutants. Emissions from one continent can influence air quality in another through an increase in the overall hemispheric burden of pollution and through discrete episodic flows of enhanced levels of pollution. This latter contribution can clearly vary with location, season, and the pollutant concerned.

There have been observations of discrete intercontinental ozone transport events made at mountain top sites and by aircraft, but the more important influence on ground-level ozone concentrations has been an apparent increase in the hemispheric burden in the troposphere. This background level is controlled primarily by global $NO_x$ emissions and to a lesser extent by methane and carbon monoxide. For aerosols and their precursors, episodic flows appear to be the most important influence. The episodes are often natural events such as dust storms, volcanic activity, or fires. For mercury, the export of emissions into the free troposphere contributes to a hemispheric or global pool of mercury. Influenced by this global pool, mercury deposition patterns are thought to be more related to patterns of emissions and precipitation than to transport events. Some persistent organic pollutants may be transported long distances as aerosols and remain where they are deposited, while others may revolatilize or become re-entrained and travel further in successive hops to reach environments far from the sources. A classic example is the presence of such pollutants in the Arctic.

Although intercontinental transport can be demonstrated using observations and measurements, it is important, not least for future policy assessment reasons, to understand how changes in emissions in one continent influence air quality in another. This requires a predictive model that can account for the nonlinearity and complexities of the atmospheric system. Ideally this simulation should be performed with an integrated system of models capable of linking the local, regional, hemispheric, and global scales. Models exist that are capable of simulating these phenomena, but more evaluation and development are required before they are applicable with confidence to policy questions.

Although current models are uncertain, the magnitudes of impacts are significant enough to suggest that further investments should be made to better characterize intercontinental transport and the potential impacts of emission changes. For ozone, it has been estimated that such transport contributes between 1 and 10 ppb to average surface ozone concentrations in Europe, North America, and Asia. Even at the low end of this range, this contribution may offset the benefits of local air pollution controls, and, depending on whether or not ozone is thought to have a threshold for adverse human health effects, it could also have important public health impacts.

For aerosols, current models and observations suggest small but significant contributions (up to 2 $\mu g/m^3$ $PM_{2.5}$ annual mean) of intercontinental transport of anthropogenic pollution. Impacts on an episodic basis can be even larger (10 to 100 $\mu g/m^3$ $PM_{10}$ maximum daily concentrations). However, such episodic events

are primarily associated with wildfires or wind-blown dust. For mercury, it has been estimated that intercontinental transport may contribute between 10 and 75% of the total deposition to the different continents in the Northern Hemisphere, with individual continents or regions providing between 1 and 40% of observed deposition. It should be stressed that these estimates are uncertain and are based on preliminary work. Further developments are under way to attempt to improve our understanding via improved descriptions of the transport processes, emission inventories, and additional observational data.

Intercontinental transport will pose significant problems not just for the scientific community but also for the policymakers. It is reassuring that the first steps in addressing the scientific issues in an institutional framework were made by the CLRTAP, which has a very strong track record in the use of science in a policy context. The Task Force on Hemispheric Transport of Air Pollution will look more broadly than the current scope and coverage of the convention, however. It will attempt to build upon a wide variety of existing research efforts in individual countries and internationally, including IGBP/IGAC, UNEP, the World Bank, GEOSS, WMO, and ACCENT within the EU. The EMEP Centres of CLRTAP will also contribute to this effort. Of particular interest is the establishment of contacts and cooperation from scientists and experts outside the UNECE regions including those from Asia, Africa, and Central America.

In the longer term, there may well be a need to integrate this scientific endeavor with policy needs. While the Task Force is at present engaged in addressing the scientific questions, thought is being given elsewhere to the policy context via the work of the Global Atmospheric Forum of IUPPA. This forum recognizes that a number of regional agreements already exist and rather than attempting to set up some form of global instrument (for example, along the lines of the Kyoto Protocol) it may be more productive, at least in the early stages of development of thinking on intercontinental transport of air pollution, to attempt instead to foster collaboration and interaction between these regional instruments. The instruments of particular importance here are the UNECE, the EU, the network of the Central Asian Republics, the Male Declaration in South Asia, the Asian Brown Cloud network, the Clean Air Initiative Asia, EANET in East Asia, APINA in Africa, IANABIS (the Inter-America Network for Atmospheric/Biospheric Studies), and the Canada/United States Air Quality Agreement. Two of the principal aims of the forum are to (1) provide a framework for dialogue and cooperation between these networks and related organizations on the practical challenges facing them and for developing joint projects and (2) to encourage harmonization of systems and approaches in key areas to facilitate cooperation at intercontinental, hemispheric, and global scales, to encourage the establishment of new networks in areas where none exist, and to encourage capacity-building in areas where resources currently constrain action.

An important area of the forum's work will also be to provide a forum for debate on common issues, and important here of course is the interaction between climate change and air pollution and the policies and institutions needed to tackle pollution at the hemispheric and global scales. This will pose challenges to policymakers and to institutions in the future. Some aspects of the issues surrounding the interaction between climate change and air quality are addressed in the next subsection.

## Air Quality and Climate Change

There are a number of emerging challenges and opportunities that are important to consider in the development of clean air policy strategies, particularly in the context of the links with climate change. Much of the integrated focus to date has been in the area of atmospheric chemistry, exploring the integration of air quality and climate change, with less emphasis on the specific emission reduction technologies and measures that will reduce emissions of both air pollutants (including air toxics) and greenhouse gases. There are also important linkages to explore between mitigation and adaptation measures, although in the latter context this is probably more for climate change than for air quality. The recently published report by Stern (2007) concluded that measures to combat climate change could lead to significant improvements in air quality and public health, citing a study by the European Environment Agency (2006a) that showed that the benefits of an emission scenario aimed at limiting global mean temperature increases to 2°C would lead to savings on the implementation of existing air pollution control measures of €10 billion per year in Europe and additional avoided health costs of €16–46 billion per year. Similarly in China, a recent study (Aunan et al., 2006) showed that for carbon dioxide reductions of 10–20%, the air pollution and other benefits more than offset the costs of action.

In North America, the integration of air issues has been somewhat constrained by a nagging ongoing debate around climate change science and whether actions to reduce GHG emissions are actually necessary, despite the findings of the Intergovernmental Panel on Climate Change, of the National Academy of Sciences, among other reputable science-based agencies and initiatives. The fact that the United States has yet to ratify the Kyoto Protocol and Canada's relatively poor record in reducing greenhouse gas emissions (currently 27% above 1990 levels, and 35% above its Kyoto target) demonstrate the lack of commitment, lack of progress, or simply the challenge of taking meaningful actions. Canada's emissions are projected to be 828 Mt by 2010 and 897 Mt by 2020 (Natural Resources Canada, 2006). Nonetheless, there is growing evidence and concern that climate change is real, is already happening (in some cases/areas occurring at a pace more quickly than previously projected), and that the impacts and effects will be severe. The Intergovernmental Panel on Climate Change (IPCC) Assessment Report Four was released in 2007, and results indicate an even greater confidence in the physical

science of climate change and the dangers that it poses to humankind (Alley et al., 2007).

Unlike the "co-benefits" research that occurred in the years leading up to the ratification of the Kyoto Protocol, the linkages between climate change and air quality will likely begin to move from the theoretical (discussion) to the practical (and applied) level more quickly than many anticipate. It is already happening in some countries. For example, the United Kingdom has analyzed the effects on carbon emissions of measures to achieve air quality objectives beyond "business as usual" in a recent review of the UK Air Quality Strategy (DEFRA, 2006) and the recent publication of the UK Climate Change program considered the impacts on pollutant emissions of the climate change measures. Some of these measures will be synergistic, leading to cumulative benefits for both air quality and climate change, but some may be contradictory, leading to conflicting outcomes.

In proposing emission standards for vehicles that will probably require aftertreatment with a small fuel economy penalty, the European Commission has tacitly accepted the small disbenefit to GHG emissions for the large improvements in particulate emissions that will result. The Council of the EU, including the United Kingdom, recently affirmed its commitment to a reduction of 20% in GHG emission reductions, and endorsed an EU objective of a 30% reduction, "provided that other developed countries commit themselves to comparable reductions and economically more advanced developing countries to contributing adequately according to their responsibilities and respective capabilities" (Council of the European Union, 2007). There is growing acceptance by environmentalists, governments, and even industry that a global reduction of 60–80% is necessary by 2050 if we are going to keep $CO_2$e (carbon dioxide equivalents) concentrations in the atmosphere below levels that would cause a 2°C increase in global temperatures (and avoid dangerous interference with the earth's climate).

With such a huge emission reduction challenge, the need to link air issues together is both necessary and unavoidable. The benefits to air quality of these longer term plans were explored in the recent UK Air Quality Strategy Review referred to earlier (see also Williams, 2007). In Canada, the current minority federal government recently introduced a Clean Air Act that addresses climate change and air quality, promising new national air quality standards and proposing GHG emission reduction targets for 2050. An absolute reduction in GHG emissions of 45 to 65% below 2006 levels by 2050 is the target. The notice of intent, however, does not lay out specific dates for new air quality standards, nor does it explain how climate change and air quality will be addressed in an integrated fashion, suggesting instead that they will be addressed as separate air issues. This is not surprising since a comparison of previous clean air strategies and climate change plans in Canada revealed that they were largely disconnected, at best providing lip service to each, but neither providing any real evidence of an integrated and coordinated approach (Bouchard, 2006).

In the fall of 2006, Rona Ambrose, who was then the Federal Minister of Environment, introduced Bill C-30 into Parliament, a new Clean Air Act (CAA) for Canada (Ambrose, 2006). Of particular concern is that the commitment to reduce GHG emissions is framed not in absolute terms, but rather in the context of allowing emissions to rise while lowering emission intensity, an approach also promoted by the Bush administration in the United States. Not surprisingly, many organizations and environmental groups have been critical of the proposed act, citing the lack of clear short-term and intermediate targets and timelines that address both criteria air contaminants and greenhouse gases. Shortly after it was announced, the act was sent to the Standing Committee on Environment and Sustainable Development for review, with members of the committee from the opposition parties determined to revise the CAA into a truly "green" piece of legislation by the end of March, 2007. Around the same time the Federal Minister of the Environment was replaced, in part reflecting her weak performance on climate change and air quality, and in anticipation of the environment becoming one of the top three issues in the next federal election. Within a broader environmental strategy, new initiatives under the CAA have also been announced targeting toxic substances, initiatives that were supported by some environmental organizations. Unfortunately, the negative reaction to the CAA as a whole was so strong that positive steps forward on toxic substances have largely been ignored. This new chemicals management strategy, announced in late 2006, will target new and existing toxic substances in Canada.

Environmentalists are also concerned that the proposed measures outlined in the CAA may turn public attention away from taking meaningful actions on climate change, since air quality had, until recent polls, been regarded as a more significant environmental issue in many parts of Canada. Indeed, the introduction of a tax credit for transit passes and proposed commitments to a 5% renewable (ethanol) content in fuels suggests that actions are more symbolic rather than aimed at making meaningful reductions in GHG emissions. Much more aggressive measures are necessary if Canada is to meet its Kyoto commitments. The latter regulation would generate a net reduction of between 2 and 4 Mt of GHGs, representing a very small dent in the 265-Mt reduction needed for Canada (from a Business As Usual projection of 828 Mt in 2010 to 563 Mt, which is 6% below the 1990 emission level of 599 Mt) to meet its Kyoto target. Furthermore, given past experience with environmental legislation (e.g., Canadian Environmental Protection Act), there are legitimate concerns that the new CAA may take years before it is passed, an unacceptable period of time in terms of the urgency to deal with climate change today, rather than tomorrow. Reductions can be achieved through the use or strengthening of existing regulatory tools, and reinventing the wheel or developing a new act is viewed largely as unnecessary.

## Science and Policy Challenges

The IPCC, in the summary for policymakers for Working Group II in the Third Assessment Report published in 2002, noted that climate change and air quality were interconnected in two key areas. First, projected climate change will be accompanied by an increase in heat waves, exacerbated by increased humidity and, in many regions, air pollution, which would lead to an increase in heat-related and smog-related deaths and illnesses (Ahmad et al., 2001). The report also notes that common air pollutants, such as nitrogen oxides, sulfur oxides, volatile organic compounds, and other particulate matter, contribute to the formation of ground-level ozone and aerosols, which have both positive and negative climate forcing. These findings suggest that as the scientific evidence of climate change and its impact on urban air quality improves, air quality management must consider the possible effects that a changing climate could have on regional air quality. Moreover, policymakers will also need to consider the additional positive or negative climate forcing that common air contaminant emissions could produce with existing or proposed air quality improvement measures.

In a more specific science and policy context, there are at least four additional aspects of the linkages between climate change and air quality that decision makers engaged in air quality management should consider:

1. The chemical/atmospheric interactions between climate change and air quality (how climate change will impact local air quality, and how air quality and emissions of particulates/aerosols affects climate change at a regional level).
2. Actions that directly reduce emissions of GHGs and other air pollutants (e.g., fuel switching, best available technologies [BATs], renewables—however some BATs that reduce air pollutants may actually increase GHG emissions).
3. Actions that indirectly reduce energy use and emissions (e.g. efficiency, conservation, pollution prevention, land use and transportation planning).
4. Actions that are both mitigation and adaptation: that is, measures that reduce emissions and reduce vulnerability by enhancing adaptive capacity (this issue has received very little attention in the climate change literature). One example is the adoption of community-based energy systems such as combined heat and power, and wind power projects that both reduce emissions and reduce vulnerability to a catastrophic systemwide failure of the energy grid.

There are also other air issues to consider, e.g., HAPs and acid rain, particularly in terms of the cumulative impacts and effects, but the primary focus for this discussion is the links between air quality and climate change. It should be noted that some sectors or regions may also be subject to emission reduction controls or targets due to nonhuman health effects, such as emissions causing acid rain, which continue to have significant impacts on aquatic and terrestrial ecosystems (Morrison & Caron, 2004).

The Air & Waste Management Association (AWMA) recently dedicated an entire issue to the linkages between air and climate change, but it was almost exclusively tied to the science and chemistry, rather than focusing on specific locations where it needs to be adopted, which sectors are more suitable for such measures, or what technologies are likely to generate the greatest benefits (Rao, 2005). A quick scan of the issue illustrates what we know so far about the science and the challenges that remain. Of note, Pennell et al. (2005) identify the following significant interactions from a scientific perspective: emissions—which are highly dependent upon future economic growth, technological change, and energy use, recognizing that we need to be moving toward a zero net carbon emissions future; atmospheric processes and effects; modeling and simulation; monitoring; and policymaking. The latter is obviously most germane to this discussion, but the authors do not go beyond concluding that the issues need to be addressed together, and that some policy analyses are beginning to focus on “co-benefits.”

“Co-benefits” are gains to the environment that accrue due to reductions in emissions of GHGs and other air pollutants, such as reduced acid rain and improved air quality, leading to improved ecosystem and human health. Such benefits from taking actions to reduce GHGs, especially by reducing fossil fuel combustion, can be significant, especially for human health, at both the global and national level. This is particularly the case for measures that reduce GHGs while at the same time reducing emissions of coarse and fine particles. A study by Davis et al. (1997) estimated that by the end of 2020, just over 700,000 premature deaths would be avoided on an annual basis, if reductions in GHG emissions of 15% and 10% for developed and developing countries were to be achieved by 2010. This included 138,000 avoided premature deaths in developed countries, of which 33,000 would be avoided in the United States alone. While Canada was not specifically considered in this study, it is expected that hundreds if not thousands of premature deaths would be avoided in this country (Chiotti & Urquizo, 2002).

Pennell et al. (2005) also note that research activities in both areas are beginning to converge, but are relatively absent in day-to-day environmental policy and technology management decisions. To complicate matters, Prinn and Dorling (2005) argue that managing air quality in a way that supports a climate-stabilization policy could be more difficult than one would think. Based on Article 2 of the United Nations Framework Convention on Climate Change, climate stabilization would occur at a level below where $CO_2$e concentrations would cause “dangerous” interference with the earth’s climate (between 450 and 550 ppmv). While this is a valid concern (and indeed a challenge), the authors provide few insights toward resolving the basic conundrum of addressing these issues together: Does addressing air quality issues through actions that reduce greenhouse gas emissions produce a broader suite of benefits and outcomes than

addressing climate change by reducing emissions of other air pollutants?

Ten years ago the challenge of addressing air quality and climate change issues together was noted by Pearce et al. (1996) in the IPCC Second Assessment Report from Working Group II. They noted that the question of secondary benefits from carbon abatement should be distinguished from the more comprehensive issue of the optimal abatement mix with respect to all pollutants. In the case of the Kyoto Protocol, the argument has largely been driven by the implicit primacy of the greenhouse problem, with improvements in air quality viewed as welcomed side effects, rather than considered in their own right. International stakeholders attending the NERAM Colloquium Series generally agree that a joint approach to the management of air quality and climate change is the best way to proceed (Craig et. al., 2007, 2008). Nonetheless, there are some who may adopt the view that perhaps each pollutant (and air issue) should be assessed (and measures adopted to reduce emissions) in proportion to the environmental damage that it causes. As Pearce et al. (1996) pointed out, interdependencies matter, as does location, and greenhouse gas emission reduction measures should be concentrated in places where the joint benefits of reducing all emissions is highest.

Similar observations are found in the United Kingdom (Air Quality Expert Group, 2007), where six key questions are raised that address four different areas related to air quality and climate change. Not surprisingly, most of the focus is on atmospheric science, and similar to the AWMA issue the report only gives cursory treatment to the implications for emissions and control options and concludes with the ominous observation that synergies and trade-offs exist in technical control measures, and that there is a need for integrated assessments across sectors and across effects. These last observations are precisely the issues that policymakers have to grapple with in the real world. They need to be recognized, explored, analyzed, and managed. There can be no hiding from them, nor denying their existence simply because they are inconvenient. As noted earlier, we are already facing some of them. With respect to the main points, the first three questions are worth closer consideration, dealing with the impact of climate change on air quality (question 1) and the impact of air quality on climate change (questions 2 and 3).

*Question 1: How could the likely impact of climate change on the general weather patterns and emissions of air pollutants and their precursors affect atmospheric dispersion and chemistry processes in general, and air quality in particular?* For example, might an increase in heatwaves affect air pollution episodes? Might the frequency and intensity of winter inversions decrease? If so, how will this affect air quality?

Several issues arise here. Unless there are any new nonlinearities introduced by enhanced climate change, the effects on policy measures are probably minor. It should simply mean that we might need to do more than we thought (more of the same) if, say, climate change leads to more frequent and more intense summer smog episodes. We might actually get more improvement than we thought, for the same emission reductions, in "winter" episodes due to less frequent and less intense winter stagnation periods. It is probable that biogenic VOC emissions will play an increasingly important role in future summer smog episodes if present warming trends continue. Emissions from these sources vary nonlinearly with temperature.

There are some estimates of climate change impacts on air quality that apply to Canada. On a broad scale, the Intergovernmental Panel on Climate Change (2001) has projected that, based on some scenarios, background levels of ground-level ozone will increase by more than 40 ppb over most mid-latitudes of the Northern Hemisphere. This would result in a doubling of average levels of ozone, and would reach levels that would be in exceedance of current Canada Wide Standards. On a finer scale, Cheng et al. (2005) provides projections for air quality affecting Windsor, Toronto, Ottawa and Montreal, and estimated that for three different emissions scenarios the number of low ground-level ozone days would generally decrease and the number of high ground-level ozone days would generally increase. In the worst-case scenario of air pollutant emissions increasing by 20% by 2050 and 32% by 2080, the study estimated that the annual total number of poor ozone days (1-h maximum $O_3 \geq 81$ ppb) could increase by 4–11 d by the 2050s, and by 10–20 d by the 2080s. The number of good d (1-h maximum $O_3 \leq 50$ ppb) could decrease by 24–40 d by the 2050s, and by 42–52 d by the 2080s.

Health Canada and Environment Canada are developing some new scenarios of climate change impacts on air quality for a national assessment on climate change and health that is due out in 2008, but it remains uncertain whether these efforts will improve our understanding substantially. It is probably prudent to agree with the Air Quality Expert Group (2007) and the point that different models show quite a wide range of responses, and that there are large uncertainties in the modeling output. In Canada, and southern Ontario and Quebec specifically, there is little doubt that air quality will get worse with climate change; however, by exactly how much is less certain, but the conclusion is nonetheless clear—that we need to do even more on reducing emissions causing air pollution, and ideally do so without adding more greenhouse gas emissions.

A logical extension of this work would be to project health effects, building on current health impact assessments. For example, an analysis of the recent summer ozone episode in the United Kingdom and Europe in August 2003 estimated that between 225 and 593 deaths were brought forward associated with ozone concentrations and some 207 associated with $PM_{10}$ (Stedman, 2004). Research in Canada has looked at the synergistic impacts of temperature change and air quality under 3 climate change emission scenarios (Cheng et al., 2005) and projected under the worst-case conditions that mortality due to poorer air quality would increase 15–25% by 2050 and from 20 to 40% by 2080. Add to these numbers projections that heat-related deaths are estimated to double and triple by 2050 and

2080 respectively. These are not trivial numbers in terms of human health, since an estimated 6,000 premature deaths already occur across Ontario due to air pollution, and hundreds of deaths due to heat stress annually (Ontario Medical Association, 2005). The synergistic and cumulative impacts on managed and unmanaged ecosystems could also be substantial, whether in concert with air quality, acid deposition, or air toxics.

Undoubtedly, a key to the assessment of future ozone effects is the issue of whether or not there is a threshold for effect. If there is, then the projected effects could be large; if there isn't, then they will be small and potentially significantly less than those due to PM.

Ozone is also difficult because the behavior of future trends depends on the metric one is examining. Peak hourly ozone will behave differently from the annual mean (in general in urban areas the former will decrease with decreasing $NO_x$ and VOCs but the latter will increase, being dominated by the titration in urban areas), and metrics between these two extremes will differ in their behavior too. In fact the behavior is controlled by three factors: (1) the local $NO_x$ environment and the titration effects; (2) the behavior of the $NO_x$/VOC smog reactions in future scenarios; and (3) the influence of the global tropospheric background. Because of these [especially (1) and (2)], future ozone trends will be strongly location specific and this means that one has to do urban scale modeling—not something the ozone modeling community has addressed very much as yet (Gower et al., 2005). This conclusion is consistent with conditions experienced in Ontario and Quebec, a region that is subject to considerable transboundary pollution from the U.S. Ohio Valley. In Toronto, for example, during smog episodes driven by ozone, more than 90% of the pollution comes from the United States, whereas during PM driven episodes, the percentage is closer to 50% (one assumes that on days when air quality levels do not warrant the issuance of a smog advisory, most of the air pollutants are from domestic sources, whether they are ozone precursors or PM) (Yap et al., 2005). This implies that local actions to reduce emissions, especially during ozone events, will have little impact on ambient conditions, and that for measures to be effective, they either have to be international or airshed in scope, or at a much finer spatial resolution. In the latter case, emission reductions targeting ozone precursors that are known to cause serious health effects (e.g., $NO_x$ and ultrafine particles) may need to be neighborhood (or even site) specific, in addition to micro-modeling of individual risk exposure. The effectiveness of site specific actions would be determined by the mix of the pollutants during the smog episode (in Ontario during the traditional smog season from May to September, although smog episodes are largely driven by ozone, they often include significant amounts of ozone precursors and PM). The other important issue in terms of climate change and ozone is the influence of biogenic emissions which needs fuller assessments in considering control scenarios for future years.

*Question 2: What are the links between the sources of emissions responsible for climate change and air quality?* What are the main scientific issues associated with the interactions of GHGs and air pollutants in the atmosphere and their impacts on climate change and air quality?

*Question 3: What do future trends in UK air pollutant emissions tell us about the potential impact on climate for the United Kingdom and Europe?* Given that some air pollutants cause air quality concerns on a regional scale, over what scale will their impact on climate be felt?

The answers to these questions are even more complex than the previous question and relationship. The role of aerosols in offsetting climate change on a local scale is still very much an emerging science, and policymakers risk venturing into the debate of having decision makers ponder polluting more $SO_2$, $NO_x$, and PM in order to offset climate change. In the Summary for Policy Makers from the IPCC Assessment Report Four, it is estimated that without the cooling effect from human-made emissions of aerosol pollutants, it is likely that greenhouse gases alone would have caused more global mean temperature rise than what was recorded in the past 50 yr (Alley et al., 2007). The report also estimates that if all sulfate aerosol particles were somehow removed from the atmosphere, there would be a rapid increase of about 0.8°C within a decade or two in the globally averaged temperature.

More specifically, the challenges, complexities, and tradeoffs between air quality and climate change can be illustrated by considering three key sectors: energy, transportation, and agriculture. Although it ultimately comes down to how society generates, produces, and uses energy, policymakers also need to consider the need to move toward a net zero carbon future, where we need to decouple the global economy from fossil fuels and rely upon noncarbon sources of energy.

Both climate change and air quality policies deal essentially with the same emission sources, so it is clearly sensible to ensure they are considered together by policymakers. Some key observations regarding challenges and opportunities are as follows:

*Energy*

- Any policy measure that reduces the use of fossil fuels in existing applications will be co-beneficial for air quality and climate change. Such measures include energy efficiency in buildings and households, which could also have co-benefits in the form of improved indoor air quality.
- Measures to increase the proportion of carbon-free energy generation in the portfolio will be co-beneficial. Sources would include wind, solar, hydro, tidal, wave, and nuclear, although some of these have their own associated problems and challenges (i.e., nuclear waste issues and public acceptance of wind farms). Community-based systems enhance local adaptive capacity, create and retain jobs in the local community, and potentially reduce a wide range of pollutants contributing to air pollution and climate change, in addition to CFCs, although these outcomes are necessarily clear (see later discussion).

- Potential trade-offs could arise where energy-generating sources fit aftertreatment of the flue gases, a practice that usually leads to a small fuel consumption penalty. Historically this penalty has been considered worth paying due to the significant air quality benefits for public health and the wider environment that can accrue. Reductions of air pollutants like sulfur dioxide have resulted in a decrease in aerosol sulfate concentrations, which, on the basis of current knowledge, has lead to an increase in radiative forcing. Despite this, it is unlikely that policy measures would be considered to increase sulfur emissions as a means of alleviating radiative forcing.
- Measures to increase the efficiency of fossil fuel use by replacing remote, central energy generation from fossil fuels by local small-scale combined heat and power sources in urban areas, running on fossil fuels or biomass, could lead to climate change/air quality trade-offs that should be quantified and assessed. Moreover, even if biomass is burned such that air quality does not worsen, but stays broadly constant, then the potentially larger air quality, public health, and environmental benefits resulting from truly zero-carbon sources of energy are foregone.

*Transport*

- The classic problem here is the diesel vehicle. This has more efficient use of fossil fuel energy than petrol/gasoline and hence smaller carbon emissions per kilometer traveled, all other things being equal. However, there are potentially significant public health disbenefits arising from the higher emissions of particulate matter that have arisen to date from diesel vehicles compared with petrol/gasoline equivalents. Technology is available to reduce significantly these emissions of particulate matter and proposals for emission standards for light-duty vehicles, which it is thought will require such technologies as have been proposed by the European Commission (see fourth subsection). Concern has been expressed that these devices lead to increased fuel consumption, on the order of a few percent, and this has been cited as a reason not to proceed with these controls. It seems likely, at the time of writing, that the EU will go ahead with agreeing to such standards so that by implication the EU has tacitly accepted that the small fuel penalty is outweighed by the relatively large (potentially of the order of about 90%) reductions in particulate matter. One factor that has not been included in analyzing these trade-offs is the benefit to radiative forcing that may arise from the reduction in black carbon emissions from diesel vehicles. The reason this has not been done is the uncertainty in the science in this area, and that of the wider issue of aerosols and climate change as a whole, where more research is clearly needed. However, measures such as particulate filters/traps that reduce particulate matter emissions by significant amounts will clearly be effective in reducing these trade-offs. As with energy use in fixed sources, any policies that lead to reduced travel and/or fuel use will be a win–win situation for climate change and air quality. Such measures are usually fiscal and could involve such policies as road user charging, fuel duty measures, tax/duty measures on high-emitting vehicles (although until the primary particulate emissions from diesel vehicles are reduced significantly—as discussed earlier—there are potential perversities in applying such measures to the current fleet). Measures on aviation are probably of wider interest in the climate change context, but reductions in $NO_x$ emissions from aircraft engines in the cruise and take-off engine modes will benefit both climate change and local air quality, which can be a problem around larger airports.
- In the medium to longer term, low carbon vehicles (hybrids, fuel cell vehicles, etc.) will also be win–wins options, providing the primary energy generation is also low or zero carbon.

*Agriculture*

- The common issues linking climate change and agriculture are mainly related to methane emissions and its impact on tropospheric ozone levels, and ammonia emissions, which can affect ecosystems directly in the vicinity of sources, and at longer range through the formation of secondary particles that can affect health and can be deposited on ecosystems where they contribute to acidification and eutrophication problems, and that also have potentially important climate effects.
- The commonality of air quality and climate change issues and tropospheric ozone is clear and there will clearly be co-benefits arising from any measures to reduce methane emissions from agricultural sources worldwide.
- Solutions to the problems of ammonia, air quality, and climate change are less obvious. The problem arises through the so-called "pollution swapping" concept and the management of nutrient nitrogen in the agricultural context. Nitrogen releases in this sector arise from fertilizer use and the excretion of nitrogen by animals. Depending on the specific local practices, residence times of manure and slurry in containers and soils, etc., this nitrogen can potentially enter the environment as nitrate in streams and rivers, where it can cause water quality problems, or it can be released to the atmosphere as

ammonia and contribute to the problems outlined already, or it can be released as nitrous oxide, a powerful greenhouse gas. Abatement methods and policies to reduce the effects of nitrogen on the environment need to recognize these potential problems and seek to find optimal solutions. This is an area of developing science.

## Future Research Requirements

What are the current gaps in our knowledge? Where should future research focus to provide appropriate scientific information to inform decisions about the comparative benefits of air quality and climate change mitigation measures? Are the currently available scientific tools sufficient to answer these gaps in our knowledge, and if not, what further developments are required? Based on the UK and Canadian experience, it may be prudent for developed and developing countries to consider the synergies between air quality and climate change policies for future time intervals, at both intermediate and long-term periods, say, 2020 and 2050. This has been done in the UK context, with specific reference to London, and the linkage between GHG emission reductions and future concentrations of $NO_x$ and PM (Williams, 2007).

At this point we can begin to at least map out the challenge. Certainly technology will play a big role in determining how successful we are in reducing emissions of GHGs and other air pollutants. In Canada, there are large dollars being invested by government and industry in sustainable technologies, and a review of the Canadian situation illustrates which sectors are attracting the most attention from an investment perspective. According to Sustainable Development Technology Canada (2007), the two sectors receiving the most funding are energy exploration and production (25%), and energy utilization (21%), followed by power generation (19%), transportation (16%), waste management (8%), forestry and wood products (6%), and agriculture (5%).

The three examples discussed earlier illustrate the challenges of making connections between air quality and climate change. These include energy—specifically the feasibility of less polluting alternatives such as "Clean Coal," green renewables (e.g., river-run hydro, wind, solar, biomass), and other less polluting fossil fuels (distributed energy systems, co-generation, natural gas, for example). The National Roundtable on the Environment and Economy recently released its own climate change and energy strategy for 2050, and this included the adoption of clean coal technologies in western Canada (in Alberta and Saskatchewan where it may be geologically feasible to sequester $CO_2$ underground, and at the same time make it easier to extract oil and natural gas from the tar sands), but this would not be suitable to Ontario where the geological conditions do not support underground storage (NRTEE, 2006). There may be some potential in the United States where coal plants are more likely to be located closer to coal mines, but under the Clean Air Interstate Rule there seems to be almost exclusive commitment to improving air quality rather than dealing with climate change (undoubtedly a reflection of the current Bush administration's attitude toward climate change), where significant reductions in $NO_x$, $SO_2$, and even mercury are possible through end-of-pipe technology. Of course these technologies do nothing about $CO_2$ and in some cases can even add GHG to the atmosphere.

This point is well known in the nonferrous smelting sector, and Inco [in October 2006 Companhia Vale do Rio Doce (CVRD) acquired control over Inco, and in November 2007 officially changed its name to Vale Inco Ltd.] and Falconbridge (in November 2006 Xtrata Plc acquired control over Falconbridge), in particular. In the case of Inco's superstack in Copper Cliff (Sudbury), Ontario, its sulfur extraction process results in a lowering of temperature in the plume that otherwise contributes to acid rain hundreds and thousands of kilometers away. The lower temperature, unless addressed, would result in the plume falling almost immediately, thereby placing Inco in a noncompliance position with respect to local air quality standards. As a result, Inco has to burn propane in order to heat up the superstack sufficiently for the $SO_2/NO_x$-laden plume to rise sufficiently high to be dispersed more broadly. Ironically, the superstack represents an outmoded 1970s technological solution to reduce pollution by dilution and to address years of noncompliance with local air standards. However, the addition of burning propane clearly puts the companies operating the smelters in conflict with any regulatory requirements or expectations to reduce GHG emissions. Furthermore, there are other smelters operating in some provinces (e.g., Manitoba) where local air quality standards and enforcement are not as strong as in Ontario, raising federal concerns about gaps in provincial regulations to protect human health. Consequently, in the recent Pollution Prevention plan posted in the *Canada Gazette* for the regulation of $SO_2$ emissions from nonferrous smelters, community-scale air standards and monitoring were also included (Department of the Environment, 2006).

Noncarbon alternatives pose challenges in terms of pricing and intermittency, and also open the door to include the nuclear option (which has resurfaced in Ontario, and in parts of Europe, despite wishes to retire and decommission nuclear plants). In many countries the best places to develop renewables, including large-scale hydro, tend to be located far away from where the demand is, which poses additional challenges in terms of transmission and distribution. Nonetheless, there is great untapped potential for renewables in many countries, including Canada. In northern Ontario, for example, untapped river-run hydro and wind power in the James Bay and Hudson's Bay lowlands could easily supplant the electricity currently generated by nuclear plants, although an extensive and costly transmission grid would need to be constructed.

It is also important to recognize that energy efficiency and energy conservation may in fact be two of our most important measures to lower emissions (by reducing the problem at the

source, simply reducing our use of energy). Canada's use of coal-fired electricity and emissions causing air pollution and climate change would be much higher today if we hadn't been so successful at improving energy intensity and efficiency. That being said, Ontario is about 50% less efficient than neighboring New York State, suggesting that there is much room for improvement (ICF, 2006). At the residential scale, energy efficiency/conservation options such as improved insulation, heat and air exchange systems, green roofs, etc. can lead to reduced energy and electricity use, and also provide co-benefits for urban biodiversity and improved indoor air quality.

In terms of transportation, improved fuel efficiency standards are essential to reduce GHG emissions, and represent an important opportunity to lower emissions causing climate change (Oliver, 2005). However, as noted in the UK experience, improved fuel efficiency does not necessarily equate to reduced air pollutant emissions (generally it does, but sometimes it does not), and an analysis of the impact on public health of the increased dieselization of the UK car fleet has recently been published (Mazzi & Dowlatabadi, 2007). Alternative fuels are also an option, as many governments are now moving toward the expansion and promotion of ethanol and biofuels. Life-cycle assessments suggest, however, that the overall benefits to the environment and health in terms of GHGs and air pollutants are not that large, if at all, depending upon the pollutant that you are considering.

Using less fuel or moving to less emitting vehicles on a kilometer-passenger basis is another option, such as modal shifts from single-occupant vehicles to public transit, carpooling, telecommuting, or active commuting. The latter has huge implications for children and youth, in terms of combating obesity and diabetes, but runs into the problem of promoting physical activity during smog episodes. Land use and transportation planning is also essential—specifically the problem of sprawl, as North American cities know only too well (to a lesser degree in Europe and Southeast Asia). In Toronto, BAU projections are for an additional 3 million people by 2030, an equal number of passenger vehicles, and a 30–40% increase in GHG emissions from transportation sources (Ontario Smart Growth, 2003). Building more efficient vehicles, installing better emission control technologies, and using alternative fuels are all good measures, but we also need to go beyond and consider not using cars period. Similar challenges exist for the movement of commercial goods and freight, involving air, rail, shipping, and intercity and local trucking. Incorporating intermodal use into a sustainable transportation strategy remains the unsolvable problem, as does addressing "just-in-time" delivery systems (the equivalent to sprawl as a huge structural problem).

In the end, it is important to recognize the need to look at these problems and challenges more closely, and to accept that these challenges are significant and that there is no silver bullet that is going to solve the problem of both air quality and climate change. A wide suite of measures will be required, and we need to move quickly and effectively. The challenge may be great, but the need to move forward in this direction is certain. As Williams (2007) states:

> We have not yet reached the limit of improvements to air quality. There will inevitably be debate over the feasibility of such improvements, and the costs which society will be prepared to devote to them. However, it seems clear that significant reductions are still possible and that such air pollution levels could represent substantial reductions in adverse effects on public health and ecosystems. There are potentially significant advantages to be gained from a harmonized and concerted analysis of policies on climate change and air quality.

## Key Messages

- The issue of air quality management is beginning to take on global dimensions, where the linkages between climate change and air pollution, how to control their sources pollutants (greenhouse gases (GHGs) and criteria air contaminants), and how they may interact to pose a cumulative risk to human health are emerging as important challenges.
- Urban areas, especially emissions and health effects associated with particulate matter (PM), are a major concern for air quality management. Other areas of concern include environmental justice and hemispheric air pollution transport.
- Adopting a risk management approach in the form of exposure-response relationships for PM is more suited for developed countries, whereas in developing countries a more traditional approach is more appropriate where recommended guidelines are expressed as a concentration and averaging time.
- For pollutants with no (or very low) effect threshold, such as $PM_{2.5}$, it will generally be more beneficial for public health to reduce pollutant concentrations across the whole of an urban area, as benefits would accrue from reductions in pollution levels even in relatively "clean" areas.
- The European Commission's adoption of an exposure reduction target, in addition to limiting the absolute maximum individual risk for European citizens, embodies a form of environmental justice, where policy measures should lead to a uniform improvement in exposure.
- Hemispheric air pollution transport poses significant challenges to the scientific community and policymakers, even at the level of local air quality management.
- The interaction between climate change and air quality poses additional challenges for policymakers. Much of the focus to date has been in the area of atmospheric chemistry, with less emphasis on specific emission reduction technologies and measures that will reduce emissions of all key pollutants (air pollutants, air toxics, and GHGs).

- Examples drawn from the EU (especially the United Kingdom) and North America (especially Canada) demonstrate the challenges of integrating climate change into the development of air quality policy strategies.
- The health benefits from integrating climate change and air quality management decisions can be nonlinear, synergistic, and in some cases counteractive. Measures must be taken that result in optimal reductions in emissions of all key pollutants, rather than at the expense of one or the other.
- Opportunities for adopting an integrated approach to air quality management include energy, transport, and agriculture. There is no silver bullet among these sectors; hence, a wide suite of effective measures will be required.

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## REFERENCES

Abbey, D. E., Nishino, N., McDonnell, W. F., Burchette, R. J., Knutsen, S. F., Beeson, W. L., and Yang, J. X. 1999. Long-term inhalable particles and other air pollutants related to mortality in nonsmokers. *Am. J. Respir. Crit. Care Med*. 159:373–382.

AEAT Environment. 2005. *Service contract for “ex-post” evaluation of short-term and local measures in the CAFE Context. A report produced for European Commission, DG Environment.* AEA Technology Environment. Available at http://ec.europa.eu/environment/air/cafe/activities/pdf/cafe_final_report.pdf.

Ahmad, Q. K., et al. 2001. Summary for policymakers. In *Climate change 2001: Working group II: Impacts, adaptation and vulnerability*, eds. J. J. McCarthy, O. F. Canziani, N. A. Leary, D. J. Dokken, and K. S. White, pp. 1–18. Cambridge: Cambridge University Press.

Air Quality Expert Group. 2007. *Air quality and climate change, A UK perspective*. Prepared for Department of Environment, Food and Rural Affairs. London. Available at www.defra.gov.uk/environment/airquality/publications/airqual-climatechange/pdf/fullreport.pdf.

Alapaty, K., Pleim, J. E., Raman, S., Niyogi, D. S., and Byun, D. W. 1997. Simulation of atmospheric boundary layer processes using local- and nonlocal closure schemes. *J. Appl. Meteorol.* 36:214–233.

Alley, R., Berntsen, T., Bindoff, N., Chen, Z., Chidthaisong, A., Friedlingstein, P., Gregory, J., Hergerl, G., Helmann, M., Hewitson, B., Hoskins, B., Joos, F., Jouzel, J., Kattsov, V., Lohmann, U., Manning, M., Matsuno, T., Molina, M., Nocholls, N., Overpeck, J., Qin, D., Raga, G., Ramaswamy, V., Ren, J., Rusticucci, M., Solomon, S., Somerville, R., Stocker, T., Stott, P., Stouffer, R., Whetton, P., Wood, R., Wratt, D., Arblaster, J., Brasseur, G., Christensen, J., Denman, K., Fahey, D., Forster, P., Jansen, E., Jones, P., Knutti, R., Le Treut, H., Lemke, P., Meehl, G., Mote, P., Randall, D., Stone, D., Trenberth, K., Willebrand, J., and F. Zwiers 2007. *Climate change 2007: The physical science basis—Summary for policy makers.* Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Available at http://www.ipcc.ch/pdf/assessment-report/ar4/wg1/ar4-wg1-spm.pdf.

Amann, M., Bertok, I., Cabala, R., Cofala, J., Heyes, C., Gyarfas, F., Klimont, Z., Schöpp, W., and Wagner, F. 2005. Target setting approaches for cost-effective reductions of population exposure to fine particulate matter in Europe. CAFE Scenario Analysis Report No. 4. Available at http://ec.europa.eu/environment/air/cafe/activities/pdf/cafe_scenario_report_4.pdf.

Ambrose, R. 2006. *Bill C-30: An Act to amend the Canadian Environmental Protection Act, 1999, the Energy Efficiency Act and the Motor Vehicle Fuel Consumption Standards Act (Canada’s Clean Air Act)*. First reading, House of Commons, Canada.

Ansari, A. S., and Pandis, S. N. 1999. An analysis of four models predicting the partitioning of semi-volatile inorganic aerosol components. *Aerosol Sci. Technol.* 31:129–153.

Aunan, K., Berntsen, T., O’Connor, D., Hindman Persson, T., Vennemo, H., and Zhai, F. 2007. Benefits and costs to China of a climate policy. *Environ. Dev. Econ.* 12: 471–497.

Avol, E. L., Gauderman, W. J., Tan, S. M., London, S. J., and Peters, J. M. 2001. Respiratory effects of relocating to areas of differing air pollution levels. *Am. J. Respir. Crit. Care Med*. 164:2067–2072.

Barton, J. 2008. Canada-wide standards and innovative transboundary air quality initiatives. *J. Toxicol. Environ. Health A* 71:74–80.

Berge, E., Huang, H. C., Chang, J., and Liu, T. H. 2001. A study of the importance of initial conditions for photochemical oxidant modeling. *J. Geophys. Res. Space.* 106:1347–1363.

Berkowitz, C. M., Fast, J. D., Springston, S. R., Larsen, R. J., Spicer, C. W., Doskey, P. V., Hubbe, J. M., and Plastridge, R. 1998. Formation mechanisms and chemical characteristics of elevated photochemical layers over the northeast United States. *J. Geophys. Res. Atmos.* 103:10631–10647.

Biswas, J., Hogrefe, C., Rao, S. T., Hao, W., and Sistla, G. 2001. Evaluating the performance of regional-scale photochemical modeling systems: Part 3. Precursor predictions. *Atmos. Environ.* 35:4175–4188.

Blanchard, C. L. 1999. Methods for attributing ambient air pollutants to emission sources. *Annu. Rev. Energy Env.* 24:329–365.

Bouchard, J. 2006. *Clean air–climate change linkages: How coherent policies can help Canada achieve emission reductions.* Master’s internship report. Concordia University, Canada.

Box, G. E. P. 1979. Robustness in the strategy of scientific model building. In *Robustness in statistics*, eds. R. L. Launer, and G. N. Wilkinson, p. 202. New York: Academic Press.

Brauer, M. Personal communication. Health and air quality: Border air quality study update. Presented at the International Airshed Strategy Co-ordinating Committee Meeting. May 31, 2006. Parksville, BC. http://www.cher.ubc.ca/UBCBAQA/Presentations/BAQS-IAS%20May31%202006.pdf

Brauer, M., Henderson, S. B, Jerrett, M., and Beckerman, B. Personal communication. Land use regression modeling of nitrogen oxides and fine particulate matter in the Greater Vancouver Regional http://www.cher.ubc.ca/UBCBAQS/BAQS%/20abstracts.pdf

Brauer, M., and Brook, J. R. 1997. Ozone personal exposures and health effects for selected groups residing in the Fraser Valley. *Atmos. Environ.* 31:2113–2121.

Brauer, M., Hoek, G., van Vliet, P., Meliefste, K., Fischer, P., Gehring, U., Heinrich, J., Cyrys, J., Bellander, T., Lewne, M., and Brunekreef, B. 2003. Estimating long-term average particulate air pollution concentrations:

Application of traffic indicators and geographic information systems. *Epidemiology* 14:228–239.

Bremauntz, A. F. 2008. Air quality management in Mexico. *J. Toxicol. Environ. Health A* 71:56–62.

Brook, J. R., and Spengler, J. D. 1995. Exposure to fine particle acidity and sulfate in 24 North American communities: The relationship between single-year observations and long-term exposures. *J. Air Waste Manage. Assoc.* 45:709–721.

Brook, J. R., Dann, T. F., and Bonvalot, Y. 1999. Observations and interpretations from the Canadian fine particle monitoring program. *J. Air Waste Manage. Assoc.* 49:35–44.

Brook, J. R., and Johnson, D. 2000. Identification of representative warm season periods for regional air quality (ozone) model simulations. *Atmos. Environ.* 34:1591–1599.

Brook, J., Dann, T., Mabury, S., and Burnett, R. 2002. *Health effects study with enhanced characterization of urban air pollutant mix*. Executive summary available at www.hc-sc.gc.ca/sr-sr/finance/tsri-irst/proj/urb-air/tsri-41_e.html.

Brook, R. D., Frankin, B., Cascio, W., Hong, Y., Howard, G., Lipsett, M., Luepker, R., Mittleman, M., Samet, J., Smith, S., and Tager, I. 2004a. Air pollution and cardiovascular disease: A statement for healthcare professionals from the expert panel on population and prevention science of the American Heart Association. *Circulation* 109:2655–2671.

Brook, J. R., Johnson, D., and Mamedov, A. 2004b. Determination of the source areas contributing to regionally high warm season $PM_{2.5}$ in eastern North America. *J. Air Waste Manage. Assoc.* 54:1162–1169.

Brook, J. R., Burnett, R. T., Dann, T. F., Cakmak, S., Goldberg, M. S., Fan, X., and Wheeler A. J. 2007a. Further interpretation of the acute effect of nitrogen dioxide observed in Canadian time series studies. *J Expo. Sci. Environ. Epidemiol.* 17:S36–S44.

Brook, J. R., Poiret, R. L., Dann. T. F., Lee, P. K. H., Lillyman, C. D., and Ip, T. 2007b. Assessing sources of $PM_{2.5}$ in cities influenced by regional transport. *J. Toxicol. Environ. Health A* 70:191–199.

Brost, R. A. 1988. The sensitivity of input parameters of atmospheric concentrations simulated by a regional chemical model. *J. Geophys. Res.* 93:2371–2387.

Brown, S. S., Ryerson, T. B., Wollny, A. G., Brock, C. A., Peltier, R., Sullivan, A. P., Weber, R. J., Dubé, W. P., Trainer, M., Meagher, J. F., Fehsenfeld, F. C., and Ravishankara, A. R. 2006. Variability in nocturnal nitrogen oxide processing and its role in regional air quality. *Science* 311:67–70.

Brunekreef, B. 1997. Air pollution and life expectancy: Is there a relation? *Occup. Environ. Med.* 54:781–784.

Brunekreef, B, Janssen, N. A., de Hartog, J., Harssema, H., Knape, M., and van Vliet, P. 1997. Air pollution from truck traffic and lung function in children living near motorways. *Epidemiology* 8:298–303.

Brunekreef, B., and Holgate, S. T. 2002. Air pollution and health. *Lancet* 360:1233–1242.

Burnett, R. T., Cakmak, S., Brook, J. R., and Krewski, D. 1997. The role of particulate size and chemistry in the association between summertime ambient air pollution and hospitalization for cardiorespiratory diseases. *Environ. Health Perspect.* 105:614–620.

Bukowiecki, N., Dommen, J., Prévôt, A. S. H., Richter, R., Weingartner, E., and Baltensperger, U. 2002. A mobile pollutant measurement laboratory—Measuring gas phase and aerosol ambient concentrations with high spatial and temporal resolution. *Atmos. Environ.* 36:5569–5579.

Burnett, R. T., Cakmak, S., Brook, J. R., and Krewski, D. 1997. The role of particulate size and chemistry in the association between summertime ambient air pollution and hospitalization for cardio-respiratory diseases. *Environ. Health Perspect.* 105:614–620.

Burnett, R. T, Stieb, D., Brook, J. R., Cakmak, S., Dales, R., Raizenne, M., Vincent, R., and Dann, T. 2004. The short-term effects of nitrogen dioxide on mortality in Canadian cities. *Arch. Environ. Health* 59:228–237.

California Air Resources Board. 2002. *The 2002 California almanac of emissions & air quality*. Sacramento: California Air Resources Board, Planning and Technical Support Division.

California Air Resources Board. 2005. *Indoor air pollution in california. Report to the california legislature*. Available at http://www.arb.ca.gov/research/indoor/ab1173/rpt0705.pdf.

Chen, I. H., Knutsen, S. F., Shavlik, D., Beeson, W. L., Petersen, F., Ghamsary, M., and Abbey, D. 2005. The association between fatal coronary heart disease and ambient particulate air pollution: Are females at greater risk? *Environ. Health Perspect.* 113:1723–1729.

Cheng, S., Campbell, M., Li, Q., Guilong, L., Auld, H., Day, N., Pengelly, D., Gingrich, S., Klaassen, J., MacIver, D., Comer, N., Mao, Y., Thompson, W., and Lin, H. 2005. *Differential and combined impacts of winter and summer weather and air pollution due to global warming on human mortality in South-Central Canada*. Technical report. Health Policy Research Program: Project no. 6795-15-2001/4400011.

Chiotti, Q., and Urquizo, N. 2002. Assessing the environment and health benefits of reducing GHG-related emissions in Canada: A discussion. *Can. J. Reg. Sci.* 25:235–257.

Chow, J. 1995. Measurement methods to determine compliance with ambient air quality standards for suspended particles. *J. Air Waste Manage. Assoc.* 45:320–382.

Chow, J. 2006. Introduction to the A & WMA 2006 critical review: Health effects of fine particulate air pollution: Lines that connect. *J. Air Waste Manage. Assoc*. 56:707–708.

City of Toronto. 2007. *Change is in the air. Toronto's commitment to an environmentally sustainable future. Framework for public review and engagement.* City of Toronto Environment Office. Available at www.toronto.ca/changeisintheair/change.htm.

Clancy, L., Goodman, P., Sinclair, H., and Dockery, D. 2002. Effect of air-pollution control on death rates in Dublin, Ireland: an intervention study. *Lancet* 360:1210–1214.

Clean Air Partnership. 2005. *A model clean air plan for the living city.* A report prepared for GTA Clean Air Council. Available at www.cleanairpartnership.org/gtacac/pdf/clean_air_plan.pdf.

Cohen, A. J., Anderson, H. R., Ostro, B., Pandey, K. D., Krzyzanowski, M., Künzli, N., Gutschmidt, K., Pope, A., Romieu, I., Samet, J. M., and Smith, K. 2005. The global burden of disease due to outdoor air pollution. *J. Toxicol. Environ. Health A* 68:1301–1307.

Cope, M. E., Hess, G. D., Lee, S., Tory, K., Azzi, M., Carras, J., Lilley, W., Manins, P. C., Nelson, P., Ng, L., Puri, K., Wong, N., Walsh, S., and Young, M. 2004. The Australian air quality forecasting system. Part I., Project description and early outcomes. *J. Appl. Meteorol.* 43:649–662.

Cote, I., Samet, J., and Vandenberg, J. 2008. U.S. Air quality management: Local, regional and global approaches. *J. Toxicol. Environ. Health A* 71: 63–73.

Council of the European Union. 2007. *Presidency conclusions of the Brussels European Council.* Available at http://www.consilium.europa.eu/ueDocs/cms_Data/docs/pressData/en/ec/93135.pdf.

Craig, L., Krewski, D., Samet, J., Shortreed, J., and van Bree, L. 2007. Strategies for clean air and health. *J. Toxicol. Environ. Health A* 70:1–4.

Craig, L., Krewski, D., Samet, J., Shortreed, J., van Bree, L., and Krupnick, A. J. 2008a. International perspectives on air quality: Risk management principles for policy development—Conference statement. *J. Toxicol. Environ. Health A* 71:4–8.

Craig, L., Krewski, D., Samet, J. M., Shortreed, J., and Kacew, S. 2008b. Colloquium statement: Strategic policy directions for air quality management. *J. Toxicol. Environ. Health* in press.

CRC. 2004. *Evaluation of the U.S. EPA MOBILE6 highway vehicle emission factor model*. Final Report, CRC Project E-64. ENVIRON International Corp., Novato, CA.

Cyrys, J., Hochadel, M., Gehring, U., Hoek, G., Diegmann, V., Brunekreef, B., and Heinrich, J. 2005. GIS-based estimation of exposure to particulate matter and $NO_2$ in an urban area: Stochastic versus dispersion modeling. *Environ. Health Perspect.* 113:987–992.

Dabberdt, W. F., Carroll, M. A., Baumgardner, D., Carmichael, G., Cohen, R., Dye, T., Ellis, J., Grell, G., Grimmond, S., Hanna, S., Irwin, J., Lamb, B., Madronich, S., McQueen, J., Meagher, J., Odman, T., Pleim, J., Schmid, H. P., and Westphal, D. 2004. Meteorological research needs for improved air quality forecasting. *Bull. Amer. Meteorol. Soc*. 85: 563–586.

Davis, D. L., Kjellstrom, T., Slooff, R., McGartland, A., Atkinson, D., Barbour, W., Hohenstein, W., Nagelhout, P., Woodruff, T., Divita, F., Wislong, J., Deck, L., and Schwartz, J. 1997. Short-term improvements in public health

from global-climate policies on fossil-fuel combustion: An interim report. *Lancet* 350:1341–1348.

Delfino, R. J., Gong, H. Jr, Linn, W. S., Pellizzari, E. D., and Hu, Y. 2003. Asthma symptoms in Hispanic children and daily ambient exposures to toxic and criteria air pollutants. *Environ. Health Perspect.* 111:647–656.

Dennis, R. L., Binkowski, F. S., Clark, T. L., McHenry, J. N., Reynolds, S., and Seilkop, S. K. 1990. Selected applications of RADM (Part II). In *NAPAP SOS/T Volume 1, Report 5 (App. 5F)*, *National Acid Precipitation Assessment Program: State of Science and Technology*, ed. P. M. Irving. Washington, DC: National Acid Precipitation Assessment Program.

Dennis, R. L., McHenry, J. N., Barchet, W. R., Binkowski, F. S., and Byun, D. W. 1993. Correcting RADM's sulfate underprediction: discovery and correction of model errors and testing the corrections through comparisons against field data. *Atmos. Environ*. 27A:975–997.

Department for Environment, Food and Rural Affairs. 2006. *The air quality strategy for England, Scotland, Wales and Northern Ireland (Vol. 1)*. Available at www.defra.gov.uk/corporate/consult/airqualstrat-review/index.htm.

Department of the Environment. 2006. Notice requiring the preparation and implementation of pollution prevention plans in respect of specified toxic substances released from base metals smelters and refineries and zinc plants, *Canada Gazette Part I* 140:17. Available at http://canadagazette.gc.ca/partI/2006/20060429/pdf/g1–14017.pdf.

Dickson, R. J., and Oliver, W. R. 1991. Emissions models for regional air quality studies. *Environ. Sci. Technol*. 25:1533–1535.

Dockery, D. W., Pope, C. A., Xu, X., Spengler, J. D., Ware, J. H., Fay, M., Ferris, B. G., and Speizer, F. E. 1993. An association between air pollution and mortality in six U.S. cities. *N. Engl. J. Med.* 329:279–285.

Dominici, F., Daniels, M., McDermott, A., Zeger, S. L., and Samet, J. M. 2003a. Shape of the exposure-response relation and mortality displacement in the NMMAPS database. In *Revised analyses of time-series studies of air pollution and health*, pp. 91–96. Boston: Health Effects Institute. Available at http://pubs.healtheffects.org/getfile.php?u=21.

Dominici, F., McDermott, A., Daniels, M., Zeger, S. L., and Samet, J. M. 2003b. Mortality among residents of 90 cities. In *Revised analyses of time-series studies of air pollution and health*, pp. 9–24. Boston: Health Effects Institute. Available at http://pubs.healtheffects.org/getfile.php?u=21.

Dominici, F., Peng, R. D., Bell, M. L., Pham, L., McDermott, A., Zeger, S. L., and Samet, J. M. 2006. Fine particulate air pollution and hospital admission for cardiovascular and respiratory diseases. *J. Am. Med. Assoc.* 295:1127–1134.

Ebelt S., Wilson W. E., and Brauer M. 2005. Exposure to ambient and nonambient components of particulate matter: a comparison of health effects. *Epidemiology* 16:396–405.

Eder, B., Kang, D., Mathur, R., Yu, S., and Schere, K. 2006. An operational evaluation of the Eta-CMAQ air quality forecast model. *Atmos. Environ*. 40:4894–4905.

Eder, B., and Yu, S. 2006. A performance evaluation of the 2004 release of Models-3 CMAQ. *Atmos. Environ*. 40:4811–4824.

EMEP. 2003. *Transboundary acidification, eutrophication and ground level ozone in Europe: Part II: Unified EMEP model performance*. Norwegian Meteorological Institute, Oslo, Norway. Available at http://www.emep.int//publ/reports/2003/emep_report_1_part2_2003.pdf.

EMEP. 2005. *Transboundary acidification, eutrophication and ground level ozone in Europe in 2003*. Norwegian Meteorological Institute, Oslo, Norway. Available at http://www.emep.int//publ/common_publications.html#2005.

Enstrom, J. E. 2005. Fine particulate air pollution and total mortality among elderly Californians, 1973–2002. *Inhal. Toxicol.* 17:803–816.

Environmental Monitoring and Reporting Branch. 2006. *Air quality in Ontario 2004 report*. Ontario Ministry of the Environment, Toronto. Available at www.ene.gov.on.ca/envision/techdocs/5383e.pdf.

EPRI. 1994. *Electric Utility Trace Substances Synthesis Report*. EPRI TR-104614. Electric Power Research Institute, Palo Alto, CA.

European Environment Agency. 2005. *Reductions in SO2 emissions from public electricity and heat production in the EU-15*. Available at http://dataservice.eea.europa.eu/atlas/viewdata/viewpub.asp?id=1796.

European Environment Agency. 2006a. *Air quality ancillary benefits of climate change policies.* EEA, Copenhagen, Denmark. Available at http://reports.eea.europa.eu/technical_report_2006_4/en/Air_quality_and_ancillary_report.pdf

European Environment Agency. 2006b. *Emissions of primary particles and secondary particulate precursors (CSI 003)*. Available at http://themes.eea.europa.eu/IMS/IMS/ISpecs/ISpecification20041001123025/IAssessment1155113516423/view_content. pp 105–136.

Evans, J., Levy, J., Hammitt, J., Santos-Burgoa, C., Castillejos, M., Caballero-Ramirez, M., Hernandez-Avila, M., Riojas-Rodriguez, H., Rojas-Bracho, L., Serrano-Trespalacios, P., Spengler, J. D., and Suh, H. 2002. Health benefits of air pollution control. In *Air quality in the Mexico megacity: An integrated assessment*, eds. L. T. Molina and M. J. Molina, pp. 105–136, Cambridge, MA: Kluwer Academic .

Finkelstein, M. M., Jerrett, M., DeLuca, P., Finkelstein, N., Verma, D. K., Chapman, K., and Sears, M. R. 2003. Relationship between income, air pollution and mortality: A cohort study. *Can. Med. Assoc. J.* 169:397–402.

Finkelstein, M. M., Jerrett, M., and Sears, M. R. 2004. Traffic air pollution and mortality rate advancement periods. *Am. J. Epidemiol.* 160:173–177.

Finkelstein M. M., Jerrett, M., and Sears, M. R. 2005. Environmental inequality and circulatory disease mortality gradients. *J. Epidemiol. Commun. Health* 59:481–486.

Fox, D. G. 1981. Judging air quality model performance. *Bull. Am. Meteorol. Soc*. 62:599–609.

Fox, D. G. 1984. Uncertainty in air quality modeling. *Bull. Am. Meteorol. Soc*. 65:27–36.

Friedman, M. S., Powell, K. E., Hutwagner, L., Graham, L. M., and Teague, W. G. 2001. Impact of changes in transportation and commuting behaviors during the 1996 summer Olympic Games in Atlanta on air quality and childhood asthma. *J. Am. Med. Assoc.* 285:897–905.

Frost, G. J., McKeen, S. A., Trainer, M., Ryerson, T. B., Neuman, J. A., Roberts, J. M., Swanson, A., Holloway, J. S., Sueper, D. T., Fortin, T., Parrish, D. D., Fehsenfeld, F. C., Flocke, F., Peckham, S. E., Grell, G. A., Kowal, D., Cartwright, J., Auerbach, N., and Habermann, T. 2006. Effects of changing power plant NOx emissions on ozone in the eastern United States: Proof of concept. *J. Geophys. Res. Atmos*. 111:D12306.

Fruin, S. A., St Denis, M. J., Winer, A. M., Colome, S. D., and Lurmann, F. W. 2001. Reductions in human benzene exposure in the California South Coast Basin. *Atmos. Environ.* 35:1069–1077.

Fung, K., Krewski, D., Burnett, R., Ramsay, T., and Chen, Y. 2005. Testing the harvesting hypothesis by time-domain regression analysis II: Covariate effects. *J. Toxicol. Environ. Health A* 68:1155–1165.

Gehring, U., Heinrich, J., Kramer, U., Grote, V., Hochadel, M., Surgiri, D., Kraft, M., Rauchfuss, K., Eberwein, H. G., and Wichmann, H. E. 2006. Long-term exposure to ambient air pollution and cardiopulmonary mortality in women. *Epidemiology* 17:545–551.

Geron, C. D., Nie, D., Arnts, R. R., Sharkey, T. D., Singsaas, E. L., Vanderveer, P. J., Guenther, A., Sickles, J. E., and Kleindienst, T. E. 1997. Biogenic isoprene emission: Model evaluation in a southeastern United States bottomland deciduous forest. *J. Geophys. Res. Atmos*. 102:18,889–18,901.

Gilbert, N., Goldberg, M. S., Beckerman, B., Brook, J. R., and Jerrett, M. 2005. Assessing spatial variability of ambient nitrogen dioxide in Montreal, Canada, with a land-use regression model. *J. Air Waste Manage. Assoc.* 55:1059–1063.

Gilliland, A. B., Appel, K. W., Pinder, R. W., and Dennis, R. L. 2006. Seasonal $NH_3$ emissions for the continental United States: Inverse model estimation and evaluation. *Atmos. Environ*. 40:4986–4998.

Gilliand, A. B., Dennis, R. L., Roselle, S. J., and Pierce, T. E. 2003. Seasonal $NH_3$ emission estimates for the Eastern United States using ammonium wet concentrations and an inverse modeling method. *J. Geophys. Res. Atmos.* 108:ACH 20-1–ACH 20-12.

Goldberg, M. S., Burnett, R., Bailar, J. C. 3rd, Brook, J., Bonvalot, Y., Tamblyn, R., Singh, R., Valois, M. F., and Vincent, R. 2001. The association between daily mortality and ambient air particle pollution in Montreal, Quebec. 2. Cause-specific mortality. *Environ. Res.* 86:26–36.

Goldberg, M. S., Burnett, R. T., Yale, J. F., Valois, M. F., and Brook, J. R. 2006. Associations between ambient air pollution and daily mortality among persons with diabetes and cardiovascular disease. *Environ. Res*. 100:255–267.

Goss, C. H., Newsom, S. A., Schildcrout, J. S., Sheppard, L., and Kaufman, J. D. 2004. Effect of ambient air pollution on pulmonary exacerbations and lung function in cystic fibrosis. *Am. J. Respir. Crit. Care Med.* 169:816–821.

Gower, S., Shortreed, J., and Chiotti, Q. 2005. *Policy analysis tools for air quality and health—Report from the May 19th, 2005 workshop*. Available at http://www.irr-neram.ca/about/May%2019%20Workshop/Policy%20Tools%20for%20AQ%20and%20Health%20-%20FINAL.pdf.

Gram, F., Nafstadb, P., and Haeheim, L. L. 2003. Estimating residential air pollution exposure among citizens in Oslo 1974–1998 using a geographical information system. *J. Environ. Monit*. 5:541–546.

Grasso, L. D. 2000. The differentiation between grid spacing and resolution and their application to numerical modeling. *Bull. Amer. Meteorol. Soc.* 81:579–580.

Gunier, R. B., Hertz., A., Von Behren, J., Reynolds, P. 2003. Traffic density in California: Socioeconomic and ethnic differences among potentially exposed children. *J. Expos. Anal. Environ. Epidemiol.* 13:240–246.

Guo, P., Xu, X., and Brook, J. R. 2006. *Development of an air quality sampling strategy for mobile measurements*. Paper 522. Air & Waste Management Association Annual Meeting, New Orleans, LA.

Haltiner, G. J., and Williams, R. T., eds. 1980. *Numerical prediction and dynamic meteorology*, 2nd ed. New York: John Wiley & Sons.

Hass, H., Builtjes, P. J. H., Simpson, D., and Stern, R. 1997. Comparison of model results obtained with several European regional air quality models. *Atmos. Environ*. 31:3259–3279.

Heald, C. L., Jacob, D. J., Fiore, A. M., Emmons, L. K., Gille, J. C., Deeter, M. N., Warner, J., Edwards, D. P., Crawford, J. H., Hamlin, A. J., Sachse, G. W., Browell, E. V., Avery, M. A., Vay, S. A., Westberg, D. J., Blake, D. R., Singh, H. B., Sandholm, S. T., Talbot, R. W., and Fuelberg, H. E. 2003. Asian outflow and trans-Pacific transport of carbon monoxide and ozone pollution: An integrated satellite, aircraft, and model perspective. *J. Geophys. Res. Atmos.* 108:ACH 25-1–ACH 25-13.

Heald, C. L., Jacob, D. J., Park, R. J., Russell, L. M., Huebert, B. J., Seinfeld, J. H., Liao, H., and Weber, R. J. 2005. A large organic aerosol source in the free troposphere missing from current models. *Geophys. Res. Lett*. 32:L18809-1–L18809-4.

Health Effects Institute. 2003. *Assessing the health Impact of air quality regulations: Concepts and methods for accountability research*. Communication 11. Boston: Health Effects Institute. Available at http://pubs.healtheffects.org/getfile.php?u=261.

Health Effects Institute. 2004. *Health effects of outdoor air pollution in developing countries of Asia: A literature review*. Special Report 15. Boston: Health Effects Institute.

Hedley, A. J., Chau, Y. K., and Wong, C. M. 2004. The change in sub-species of particulate matter ($PM_{10}$) before and after an intervention to restrict sulphur content of fuel in Hong Kong. Paper presented at Better Air Quality (BAQ) conference, Agra, India, 6–8 December 2004.

Hedley, A. J., McGhee, S. M., Barron, B., Chau, J., Thach, T. Q., Wong, T. W., Loh, C., and Wong, C.-M. Personal communication. Air pollution in Hong Kong and the Pearl River Delta. Paper presented at NERAM Colloquium V. Strategic Policy Directions for Air Quality Management. October 16, 2006. Vancouver, BC.

Hedley, A. J., McGhee, S. M., Barron, B., Chau, Y. K., Chau, J., Thach, T. Q., Wong, T. W., Loh, C., and Wong, C. M. 2008 Air Pollution: costs and paths to a sollution in Hong Kong-Understanding the connection between visibility, air pollution and health costs in pursuit of accountability, environmental justice and health. *J. Toxicol. Environ. Health A* in press.

Hedley, A. J., Wong, C. M., Thach, T. Q., Ma., S. Lam, T. H., and Anderson, H. R. 2002. Cardiorespiratory and all-cause mortality after restrictions on sulphur content of fuel in Hong Kong: An intervention study. *Lancet* 360:1646–1652.

Hodzic, A., Vautard, R., Chazette, P., Menut, L., and Bessagnet, B. 2006. Aerosol chemical and optical properties over the Paris area within ESQUIF project. *Atmos. Chem. Phys. Discuss*. 6:401–454.

Hoek, G., Brunekreef, B., Goldbohm, S., Fischer, P., and van den Brandt, P. A. 2002. Association between mortality and indicators of traffic-related air pollution in the Netherlands: A cohort study. *Lancet* 360:1203–1209.

Hogrefe, C., Rao, S. T., Kasibhatla, P., Kallos, G., Tremback, C. J., Hao, W., Olerud, D., Xiu, A., McHenry, J., and Alapaty, K. 2001a. Evaluating the performance of regional-scale photochemical modeling systems: Part 1. Meteorological predictions. *Atmos. Environ*. 35:4159–4174.

Hogrefe, C., Rao, S. T., Kasibhatla, P., Hao, W., Sistla, G., Mathur, R., and McHenry, J. 2001b. Evaluating the performance of regional-scale photochemical modeling systems: Part 2. Ozone predictions. *Atmos. Environ*. 35:4175–4188.

Hogrefe, C., Sistla, G., Zalewsky, E., Hao, W., and Ku, J.-Y. 2003. An assessment of the emissions inventory processing systems EMS-2001 and SMOKE in grid-based air quality models. *J. Air Waste Manage. Assoc*. 53:1121–1129.

Houyoux, M. R., Vukovich, J. M., Coats, C. J., Jr., and Wheeler, N. J. M. 2000. Emission inventory development and processing for the Seasonal Model for Regional Air Quality (SMRAQ) project. *J. Geophys. Res. Atmos.* 105:9079–9090.

ICF Consulting. 2006. *The electricity supply/demand gap and the role of efficiency and renewables in Ontario.* Toronto, Ontario. Available at http://www.pollutionprobe.org/Reports/elec_supplydemandICF.pdf.

Intergovernmental Panel on Climate Change. 2001. Question 8. In *Climate change 2001: Synthesis report. A contribution of working groups I, II and III to the third assessment report of the Intergovernmental Panel on Climate Change*, eds. R. T. Watson and the Core Writing Team, pp. 123–134. Cambridge: Cambridge University Press. Available at www.grida.no/climate/ipcc_tar/vol4/english/ 060.htm.

Ito, K. 2003. Associations of particulate matter components with daily mortality and morbidity in Detroit, Michigan. In *Revised analyses of time-series studies of air pollution and health*, pp. 143–156. Boston: Health Effects Institute. Available at http://pubs.healtheffects.org/getfile.php?u=21.

Jacobson, M. Z. 1999. *Fundamentals of Atmospheric Modeling*. Cambridge, UK: Cambridge University Press.

Janssen, N. A. J., Brunekreef, B., van Vliet, P., Aarts, F., Meliefste, K., Harssema, H., and Fischer, P. 2003. The relationship between air pollution from heavy traffic and allergic sensitization, bronchial hyperresponsiveness, and respiratory symptoms in Dutch schoolchildren. *Environ. Health Perspect.* 111:1512–1518.

Jerrett, M., Burnett, R. T., Ma, R., Pope, C. A., Krewski, D., Newbold, K. B., Thurston, G., Shi, Y., Finkelstein, N., Calle, E. E., and Thun, M. J. 2005. Spatial analysis of air pollution and mortality in Los Angeles. *Epidemiology* 116: 727–736.

Jerrett, M., Newbold, K. B., Burnett, R. T., Thurston, G., Lall, R., Pope, C. A. III., Ma, R., DeLuca, P., Thun, M. J., Calle, E. E., and Krewski, D. 2007a. Geographies of uncertainty in the health benefits of air quality improvements. *Stoch. Environ. Res. Risk Assess.* 21:511–522.

Jerrett, M., Arain, M. A., Kanaroglou, P., Beckerman, B., Crouse, D., Gilbert, N. L., Brook, J. R., and Finkelstein, N. 2007b. Modelling the intra-urban variability of ambient traffic pollution in Toronto, Canada. *J. Toxicol. Environ. Health A* 70:200–212.

Kanoraglou, P. S., Jerrett, M., Morrison, J., Beckerman, B., Arain, M. A., Gilbert, N. L., and Brook, J. R. 2005. Establishing an air pollution monitoring network for intra-urban population exposure assessment: a location-allocation approach. *Atmos. Environ*. 39:2399–2409.

Karamchandani, P., and Venkatram, A. 1992. The role of non-precipitating clouds in producing ambient sulfate during summer: Results from simulations with the Acid Deposition and Oxidant Model (ADOM). *Atmos. Environ.* 26A:1041–1052.

Kim, D., Sass-Kortsak, A., Purdham, J. T., Dales, R. E., and Brook, J. R. 2006. Associations between personal exposures and fixed-site ambient measurements of fine particulate matter, nitrogen dioxide and carbon monoxide in Toronto, Canada. *J. Expos. Anal. Environ. Epidemiol.* 15:172–183.

Kleeman, M. J., and Cass, G. R. 1999a. Identifying the effect of individual emissions sources on particulate air quality within a photochemical aerosol processes trajectory model. *Atmos. Environ*. 33:4597–4613.

Kleeman, M. J., and Cass, G. R. 1999b. Effects of emissions control strategies on the size and composition distribution of urban particulate air pollution. *Environ. Sci. Technol*. 33:177–189.

Kleinman L. I. 1987. Source-receptor relations from nonlinear atmospheric models, *Atmos. Environ*. 21:1219–1225.

Knipping, E. M., Kumar, N., Pun, B. K., Seigneur, C., Wu, S.-Y., and Schichtel, B. A. 2006. Modeling regional haze during the BRAVO study using CMAQ-MADRID: 2. Source region attribution of particulate sulfate compounds. *J. Geophys. Res. Atmos.* 111:D06303.

Kolb, C. E., Herndon, S. C., McManus, J. B., Shorter, J. H., Zahniser, M. S Nelson, D. D. Jayne, J. T. Canagaratna, M. R., and Worsnop, D. R.

2004. Mobile laboratory with rapid response instruments for real-time measurements of urban and regional trace gas and particulate distributions and emission source characteristics. *Environ. Sci. Technol.* 38: 5694–5703.

Krewski, D., Burnett, R. T., Goldberg, M. S., Hoover, K., Siemiatycki, J., Jerrett, M., Abrahamowicz, M., and White, W. H. 2000. *Reanalysis of the Harvard six cities study and the American Cancer Society study of particulate air pollution and mortality*. Cambridge, MA: Health Effects Institute. Available at http://pubs.healtheffects.org/view.php?id=6.

Krewski, D., Burnett, R., Goldberg, M. S., Hoover, B. K., Siemiatycki, J., Abrahamowicz, M., and White, W. H. 2004. Validation of the Harvard six cities study of air pollution and mortality. *N. Engl. J. Med.* 350:198–199.

Krewski, D., Burnett, R., Goldberg, M. S., Hoover, B. K., Siemiatycki, J., Abrahamowicz, M., and White, W. H. 2005a. Reanalysis of the Harvard six cities study, Part I: Validation and replication. *Inhal. Toxicol.* 17:335–342.

Krewski, D., Burnett, R., Goldberg, M. S., Hoover, B. K., Siemiatycki, J., Abrahamowicz, M., Villeneuve, P. J., and White, W. H. 2005b. Reanalysis of the Harvard six cities study, Part II: Sensitivity analysis. *Inhal. Toxicol.* 17:343–353.

Krupnick, A. J. 2008. Challenges to managing air pollution. *J. Toxicol. Environ. Health A* 71:13–23.

Kuhlbusch, T., Astrid, J., Hugo, A., Peters, A. von Klot, S., Cyrys, J., Wichmann, H.-E., Quass, U., and Bruckmann, P. 2004. *Analysis and design of local air quality measurements: Towards European Air Quality Health Effect Monitoring*. Available at http://ec.europa.eu/environment/air/cafe/activities/pdf/euraqhem_final_report.pdf.

Kuhn, M., Builtjes, P. J. H., Poppe, D., Simpson, D., Stockwell, W. R., Andersson-Skold, Y., Baart, A., Das, M., Fiedler, F., Hov, Ø, Kirchner, F., Makar, P. A., Milford, J. B., Roemer, M. G. M., Ruhnke, R., Strand, A., Vogel, B., and Vogel, H. 1998. Intercomparison of the gas-phase chemistry in several chemistry and transport models. *Atmos. Environ*. 32:693–709.

Laden, F., Schwartz, J., Speizer, F. E., and Dockery, D. W. 2006. Reduction in fine particulate air pollution and mortality: Extended follow-up of the Harvard six cities study. *Am. J. Respir. Crit. Care Med*. 73:667–672.

Lipfert, F. W., Wyzga, R. E., Baty, J. D., and Miller, J. P. 2006. Traffic density as a surrogate measure of environmental exposures in studies of air pollution health effects: Long-term mortality in a cohort of U.S. veterans. *Atmos. Environ.* 40:154–169.

Liu, Y., Sarnat, J. A., Kilaru, V., Jacob, D. J., Li, Q., and Koutrakis, P. 2005. Estimating ground-level $PM_{2.5}$ in the eastern United States using satellite remote sensing. *Environ. Sci. Technol*. 39:3269–3278.

Logan, J. A. 1999. An analysis of ozonesonde data for the troposphere: Recommendations for testing 3-D models and development of a gridded climatology for tropospheric ozone. *J. Geophys. Res. Atmos*. 104:16, 115–16,149.

Luginaah, I., Xu, X., Fung, K. Y., Grgicakmannion, A., Wintermute, J., Wheeler, A. A., and Brook, J. 2006. Establishing the spatial variability of ambient nitrogen dioxide in Windsor, Ontario. *Int. J. Environ. Stud.* 63:487–500.

Makar, P. M., Wiebe, H. A., Staebler, R. M., Li, S.-M., and Anlauf, K. G. 1998. Measurement and modeling of nighttime particle nitrate formation. *J. Geophys. Res. Space*. 103:13,095–13,110.

Mallet, V., and Sportisse, B. 2005. A comprehensive study of ozone sensitivity with respect to emissions over Europe with a chemistry-transport model. *J. Geophys. Res. Atmos.* 110:1–15.

Mallet, V., and Sportisse, B. 2006. Uncertainty in a chemistry-transport model due to physical parameterizations and numerical approximations: An ensemble approach applied to ozone modeling. *J. Geophys. Res. Atmos.* 111:D01302.

Manins, P. C., Cope, M. E., Hurley, P. J., Newton, P. W., Smith, N. C., and Marquez, L. O. 1998. *The impact of urban development on air quality and energy use*. Proc. 14th International Clean Air & Environment conference, Melbourne, Australia.

Marmur, A., Park, S.-K., Mulholland, J. A., Tolbert, P. E., and Russell, A. G. 2006. Source apportionment of $PM_{2.5}$ in the southeastern United States using receptor and emissions-based models: Conceptual differences and implications for time-series health studies. *Atmos. Environ*. 40:2533–2551.

Marshall, J. D., Riley, W. J., McKone, T. E., and Nazaroff, W. W. 2003. Intake fraction of primary pollutants: motor vehicle emissions in the South Coast Air Basin. *Atmos. Environ* 37:3455–3468.

Martin, R. V., Fiore, A. M., and Van Donkelaar, A. 2004. Space-based diagnosis of surface ozone sensitivity to anthropogenic emissions. *Geophys. Res. Lett.* 31:L06120–1–L06120–4.

Martin, R. V., Jacob, D. J., Chance, K., Kurosu, T. P., Palmer, P. I., and Evans, M. J. 2002. Global inventory of nitrogen oxide emissions constrained by space-based observations of $NO_2$ columns. *J. Geophys. Res. Atmos.* 108:ACH 5–1–ACH 5–12.

Maynard, R. 2003a. Scientific information needs for regulatory decision making. *J. Toxicol. Environ. Health A* 66:1499–1501.

Maynard, R., Krewski, D., Burnett, R., Samet, J., Brook, J., Granville, G., and Craig, L. 2003b. Health and air quality: Directions for policy-relevant research. *J. Toxicol. Environ. Health A* 66:1891–1903.

Mazzi, E. A., and Dowlatabadi, H. 2007. Air quality impacts of climate mitigation: UK policy and passenger vehicle choice. *Environ. Sci. Technol*. 41:387–392.

McHenry, J. N., Binkowski, F. S., Dennis, R. L., Chang, J. S., and Hopkins, D. 1992. The Tagged Species Engineering Model (TSEM). *Atmos. Environ*. 26A:1041–1052.

McKeen, S., Wilczak, J., Grell, G., Djalalova, I., Peckham, S., Hsie, E.-Y., Gong, W., Bouchet, V., Ménard, S., Moffet, R., McHenry, J., McQueen, J., Tang, Y., Carmichael, G. R., Pagowski, M., Chan, A., and Dye, T. 2005. Assessment of an ensemble of seven real-time ozone forecasts over Eastern North America during the summer of 2004. *J. Geophys. Res. Atmos*. 110:1–16.

McNair, L. A., Harley, R. A., and Russell, A. G. 1996. Spatial inhomogeneity in pollutant concentrations, and their implications for air quality model evaluations. *Atmos. Environ*. 30:4291–4301.

Ménard, R. Personal communication. Chemical-dynamic coupling in data assimilation. Presented at Air Quality 2006: Piecing the Puzzle. The Fourth Canadian Workshop on Air Quality. Feb. 16, 2006. Banff, AB.

Ménard, R., and Robichaud, A. 2005. *The chemistry-forecast system at the meteorological service of Canada*. Meteorological Service of Canada, Dorval, QC, Canada. Available at www.ecmwf.int/publications/library/ecpublications/_pdf/seminar/2005/sem2005_menard.pdf.

Mendoza-Dominguez, A., and Russell, A. G. 2001. Estimation of emission adjustments from the application of four-dimensional data assimilation to photochemical air quality modeling. *Atmos. Environ*. 35:2879–2894.

Meng, Z., Dabdub, D., and Seinfeld, J. H. 1997. Chemical coupling between atmospheric ozone and particulate matter. *Science* 277:116–119.

Mexico Air Quality Management Team. 2002. *Improving air quality in metropolitan Mexico city: An economic valuation.* The World Bank, Latin America and the Caribbean region Environmentally and Socially Sustainable Development Sector Unit, Washington, DC. Available at www.idrc.ca/IMAGES/ecohealth/100205-3.pdf.

Miller, C. A., Hidy, G., Hales, J., Kolb, C. E., Werner, A. S., Haneke, B., Parrish, D., Frey, C., Rojas-Bracho, L., Deslauriers, M., Pennell, B., and Mobley, J. D. 2006. Air emission inventories in North America: A critical assessment, *J. Air Waste Manage. Assoc.* 56:1115–1129.

Misra, P. K., Bloxam, R., Fung, C., and Wong, S. 1989. Non-linear response of wet deposition to emissions reduction: A model study. *Atmos. Environ*. 23:671–687.

MIT Laboratory for Energy and the Environment. 2002. Air quality management in the Mexico City metropolitan area: An interview with Mario Molina and Luisa Molina. *Initiatives Energy Environ.* 4:8–9. Available at http://lfee.mit.edu/public/LFEEnews_4.2.pdf.

Molina, L. T., and Molina, M. J., eds., 2002. A*ir quality in the Mexico megacity: An integrated assessment*. Cambridge, MA: Kluwer Academic.

Molina, L. T., Molina, M. J., Slott, R. S., Kolb, C. E., Gbor, P. K., Meng, F., Singh, R. B., Galvez, O., Sloan, J. J., Anderson, W. P., Tang X. Y., Hu, M., Xie, S., Shao, M., Zhu, T., Zhang, Y. H., Gurjar, B. R., Artaxo, P. E., Oyola, P., Gramsch, E., Hidalgo, D., and Gertler, A. W. 2004. *2004 Critical review on megacities and atmospheric pollution: Air quality in selected megacities* [online version]. Available at http://secure.awma.org/JOURNAL/pdfs/2004/12/onlineversion2004.PDF.

Molina, M. J., and Molina, L. T. 2004a. Megacities and atmospheric pollution. *J. Air Waste Manage. Assoc.* 54:644–680.

Molina, L. T., and Molina, M. J. 2004b. Improving air quality in megacities: Mexico case study. *Ann. NY Acad. Sci.* 1023:142–158.

Moran, M. D. 2000. Basic aspects of mesoscale atmospheric dispersion. In *Mesoscale atmospheric dispersion*, ed. Z. Boybeyi, pp. 27–119. Southampton, UK: WIT Press.

Moran, M. D., and Zheng, Q. 2006. *Modelling long-term sulfur and nitrogen deposition using Lagrangian chemical transport models*. Proc. 14th Joint AMS/A&WMA Conference on the Applications of Air Pollution Meteorology, Atlanta. Available at http://ams.confex.com/ams/pdfpapers/101260.pdf.

Moran, M. D. 2005. Current and proposed emission control programs: How will acid deposition be affected? In *2004 Canadian acid deposition science assessment*, pp. 99–162. Downsview, ON: Environment Canada.

Morrison, H., and Caron, S., eds. 2004. *2004 Canadian acid deposition science assessment*. Downsview, ON: Environment Canada.

NARSTO. 2004. *Particulate matter science for policy makers: A NARSTO assessment*, eds. P. McMurry, M. Shepherd, and J. Vickery, pp. 235–281. Cambridge: Cambridge University Press.

NARSTO. 2005. *Improving emission inventories for effective air quality management across North America: A NARSTO assessment*. NARSTO-05-001. Oak Ridge, TN.

National Research Council. 2004. *Air quality management in the United States.* Washington, DC: National Academies Press.

Natural Resources Canada. 2006. *Canada's energy outlook: The reference case 2006*. Natural Resources Canada, Ottawa, Canada. Available at http://www.nrcan.gc.ca/com/resoress/publications/peo/peo2006-eng.pdf.

National Round Table on the Environment and the Economy. 2006. *Advice on a long-term strategy on energy and climate change.* Available at http://www.nrtee-trnee.ca/eng/publications/wedge-advisory-note/ecc-wedge-advisory-note-eng.pdf.

Odum, J. R., Hoffmann, T., Bowman, F., Collins, D., Flagan, R. C., and Seinfeld, J. H. 1996. Gas/particle partitioning and secondary organic aerosol yields. *Environ. Sci. Technol*. 30:2580–2585.

O'Connor, S., and Cross, R. 2006. California's achievements in mobile source emission control. *EM: Air Waste Manage. Assoc. Mag. Environ. Managers* July:28.

Official Journal of the European Communities. 1996. *Council Directive 96/62/EC on Ambient air quality assessment.* No. L 296/55. Available at http://eur-lex.europa.eu/LexUriServ/LexUriS-erv.do?uri=CELEX:31996L0062:EN:HTML.

Official Journal of the European Communities. 1999. *Council Directive 1999/30/EC relating to limit values for sulfur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air*. No. L 163/41.

Oliver, B. 2005. *Greenhouse gas emissions and vehicle fuel efficiency standards for Canada.* Toronto: Pollution Probe. Available at www.pollutionprobe.org/Reports/vehiclefuel.pdf.

O'Neill, M. S., Veves, A., Zanobetti, A., Sarnat, J. A., Gold, D. R., Economides, P. A., Horton, E. S., and Schwartz, J. 2005. Diabetes enhances vulnerability to particulate air pollution-associated impairment in vascular reactivity and endothelial function. *Circulation* 111:2913–2920.

Ontario Medical Association. 2005. *The illness costs of air pollution. 2005–2026 Health and economic damage estimates*. Available at http://www.oma.org/Health/smog/report/ICAP2005_Report.pdf.

Ontario Smart Growth. 2003. *Shape the future: Central Ontario smart growth panel*. Final report. Toronto: Government of Ontario.

Oreskes, N., Shrader-Frechette, K., and Belitz, K. 1994. Verification, validation, and confirmation of numerical models in the earth sciences. *Science* 263:641–646.

Padro, J., Puckett, K. J., and Woolridge, D. N. 1993. The sensitivity of regionally averaged $O_3$ and $SO_2$ concentrations to ADOM dry deposition velocity parameterizations. *Atmos. Environ*. 27A:2239–2242.

Palacios, M., Kirchner, F., Martilli, A., Clappier, A., Martín, F., and Rodríguez, M. E. 2002. Summer ozone episodes in the Greater Madrid area. Analyzing the ozone response to abatement strategies by modelling. *Atmos. Environ*. 36:5323–5333.

Palmer, P. I., Jacob, D. J., Jones, D. B. A., Heald, C. L., Yantosca, R. M., Logan, J. A., Sachse, G. W., and Streets, D. G. 2003. Inverting for emissions of carbon monoxide from Asia using aircraft observations over the western Pacific. *J. Geophys. Res. Atmos.* 108:GTE 49–1–GTE 49–12.

Park, R. J., Jacob D. J., Chin, M., and Martin, R. V. 2003. Sources of carbonaceous aerosols over the United States and implications for natural visibility. *J. Geophys. Res. Atmos.* 108:AAC 5–1–AAC 5–14.

Parrish, D. D., Trainer, M., Hereid, D., Williams, E. J., Olszyna, K. J., Harley, R. A., Meagher, J. F., and Fehsenfeld, F. C. 2002. Decadal change in carbon monoxide to nitrogen oxide ratio in U.S. vehicular emissions. *J. Geophys. Res.-Atmos.* 107:ACH 5–1–ACH 5–9.

Pearce, D. W., Cline, W. R., Anchanta, A. N., Frankhauser, S., Pachauri, R. K., Toll, R. S. J., and Vellinga, P. 1996. The social costs of climate change: Greenhouse damage and the benefits of control. In *Climate change 1995: Economic and social dimensions of climate change: Contributions of working group III to the second assessment report change*, eds. J. P. Bruce, L. Hoesung, and E. F. Haites, pp. 179–224. New York: Cambridge University Press.

Pengelly, L. D., and Sommerfreund, J. 2004. *Air pollution-related burden of illness in Toronto: 2004 update*. Toronto: Environmental Protection Office, Toronto Public Health.

Peters, L. K., Berkowitz, C. M., Carmichael, G. R., Easter, R. C., Fairweather, G., Ghan, S. J., Hales, J. M., Leung, L. R., Pennell, W. R., Potra, F. A., Saylor, R. D., and Tsang, T. T. 1995. The current state and future direction of Eulerian models in simulating the tropospheric chemistry and transport of trace species: A review. *Atmos. Environ*. 29:189–222.

Pennell, W., Scheraga, J., Rao, S. T., and Foley, G. 2005. Air quality and climate change: dual challenges for the 21st century. *EM: Air Waste Manage. Assoc. Mag. Environ. Managers* October:8.

Peters, J., Hedley, A. J., Wong, C. M., Lam, T. H., Ong, S. G., Liu, J., and Spiegelhalter, D. J. 1996. Effects of an ambient air pollution intervention and environmental tobacco smoke on children's respiratory health in Hong Kong. *Int. J. Epidemiol.* 25:821–828.

Peters, J. M., Avol, E., Navidi, W., London, S. J., Gauderman, W. J., Lurmann, F., Linn, W., Margolis, H., Rappaport, E., Gong, H., Jr., and Thomas, D. C. 1999. A study of twelve Southern California communities with differing levels and types of air pollution. *Am. J. Respir. Crit. Care Med.* 159:760–767.

Petron, G., Granier, C., Khattatov, B., Lamarque, J.-F., Yudin, V., Muller, J.-F., and Gille, J. 2002. Inverse modeling of carbon monoxide surface emissions using CMDI network observations. *J. Geophys. Res.* 107:4761.

Pielke, R. A. 1984. *Mesoscale meteorological modeling.* Orlando, FL: Academic Press.

Pielke, R. A. and Uliasz, M. 1998. Use of meteorological models as input to regional and mesoscale air quality models—Limitations and strengths. *Atmos. Environ*. 32:1455–1466.

Pierce, T., Geron, C., Bender, L., Dennis, R., Tonnesen, G., and Guenther, A. 1998. Influence of increased isoprene emissions on regional ozone modeling. *J. Geophys. Res. Atmos*. 103:25,611–25,629.

Pleim, J. E., and Xiu, A. 1995. Development and testing of a surface flux and planetary boundary layer model for application in mesoscale models. *J. Appl. Meteor*. 34:16–32.

Polina, B. M., Offenberg, J. H., Clemente, J., Blaustein, M., Thurston, G. D., and Chen, L. C. 2004. Ambient pollutant concentrations measured by a mobile laboratory in South Bronx, NY. *Atmos. Environ.* 38:5283–5294.

Pope, C. A. 1989. Respiratory disease associated with community air pollution and a steel mill, Utah Valley. *Am. J. Public Health* 79:623–628.

Pope, C. A., and Dockery, D. W. 2006. Health effects of fine particulate air pollution: Lines that connect. 2006 Critical review. *J. Air Waste Manage. Assoc*. 56:709–742.

Pope, C. A., Thun, M. J., Namboodiri, M. M., Dockery, D. W., Evans, J. S., Speizer, F. E., and Heath, C. W. 1995. Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults. *Am. J. Respir. Crit. Care Med.* 151:669–674.

Pope, C. A., Burnett, R. T., Thun, M. J., Calle, E. E., Krewski, D., Ito, K., and Thurston, G. D. 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *J. Am. Med. Assoc.* 287:1132–1141.

Pope, C. A., Burnett, R. T., Thurston, G. D., Thun, M. J., Calle, E. E., Krewski, D., and Godleski, J. J. 2004. Cardiovascular mortality and long term exposure to particulate air pollution. *Circulation* 109:71–77.

Prinn, R., and Dorling, S. 2005. Climate change and air quality: international perspective and policy implications. *EM: Air Waste Manage. Assoc. Mag. Environ. Managers* October:37.

Proctor, S. D., Dreher, K. L., Kelly, S. E., and Russell, J. C. 2006. Hypersensitivity of prediabetic JCR: LA-cp rats to fine airborne combustion particle-induced direct and noradrenergicmediated vascular contraction. *Toxicol. Sci.* 90:385–391.

Pudykiewicz, J. A. 1998. Application of adjoint tracer transport equations for evaluating source parameters. *Atmos. Environ*. 32:3039–3050.

Pun, B. K., and Seigneur, C. 1999. Understanding particulate matter formation in the California San Joaquin Valley: Conceptual model and data needs. *Atmos. Environ*. 33:4865–4875.

Pun, B. K., Seigneur, C., Vijayaraghavan, K., Wu, S.-Y., Chen, S.-Y., Knipping, E. M., and Kumar, N. 2006. Modeling regional haze in the BRAVO study using CMAQ-MADRID: 1. Model evaluation. *J. Geophys. Res. Atmos*. 111:D06302.

Raizenne, M. 2003. Science and regulation—U.S. and Canadian overview. *J. Toxicol. Environ. Health A* 66:1503–1506.

Rao, S. T. 2005. Understanding the interactions between climate change and air quality. *EM: Air Waste Manage. Assoc. Mag. Environ. Managers* October:6.

Reid, N., Misra, P. K., Amman, M., and Hales, J. 2003. *Air quality modelling. Paper presented at Third NERAM International Colloquium on Health and Air Quality. 5–7 November, 2003*. Santo Spirito Hospital, Rome, Italy. Available at www.irr-neram.ca/rome/rome.html#proceed.

Rosenlund, M., Berglind, N., Pershagen, G., Hallqvist, J., Jonson, T., and Bellander, T. 2006. Long-term exposure to urban air pollution and myocardial infarction. *Epidemiology* 17:383–390.

Russell, A., and Dennis, R. 2000. NARSTO critical review of photochemical models and modeling. *Atmos. Environ*. 34:2283–2324.

Sahsuvaroglu, T., Arain, A., Kanaroglou, P., Finkelstein, N., Newbold, B., Jerrett, M., Beckerman, B., Brook, J., Finkelstein, M., and Gilbert, N. L. 2006. A land-use regression model for predicting ambient concentrations of nitrogen dioxide in Hamilton, Canada. *J. Air Waste Manage. Assoc.* 56:1059–1069.

Samaniego, J., and Figueres, C. 2002. Evolving to a sector-based clean development mechanism. In *Building on the Kyoto protocol: Options for protecting the climate*, eds. K. A. Baumert, O. Blanchard, S. Llosa, and J. Perkaus, pp. 89–108. Washington, DC: World Resources Institute.

Samet, J. Personal communication. Impact of air pollution on public health. Transportaability of risk estimates. Presentation at NERAM Colloquium V: Strategic Policy Decision for Air Quality Management. October 16, 2006. Vancouver BC.

Samoli, E., Aga, E., Toulouni, G., Nisictes, K., Forsberg, B., Lefranc, A., Pekhanen, J., Wojtynich, B., Schindler, C., Niciu, E., Brunstein, R., Dodic Fihfch, M., Schwartz, J., and Katsoujanni, K. 2006. Short term effects of nitrogen dioxide and mortality: An analysis within the APHEA project. *Eur. Respir. J.* 27:1129–1137.

Sarnat, J. A., Schwartz, J., Catalano, P. J., and Suh, H. H. 2001. Gaseous pollutants in particulate matter epidemiology: Confounders or surrogates? *Environ. Health Perspect.* 10:1053–1061.

Schoeberl, M. R., Douglass, A. R., Hilsenrath, E., Bhartia, P. K., Barnett, J., Gille, J., Beer, R., Gunson, M., Waters, J., Levelt, P. F., and DeCola, P. 2004. Earth observing system missions benefit atmospheric research. *EOS* 85:177–178.

Schwartz, J. 2000. Harvesting and long term exposure effects in the relation between air pollution and mortality. *Am. J. Epidemiol.* 151:440–448.

Seaman, N. L. 2000. Meteorological modeling for air-quality assessments. *Atmos. Environ*. 34:2231–2259.

Seigneur, C. 2001. Current status of air quality models for particulate matter. *J. Air Waste Manage. Assoc*. 51:1508–1521.

Seigneur, C., Pun, B., Pai, P., Louis, J.-F., Solomon, P., Emery, C., Morris, R., Zahniser, M., Worsnop, D., Koutrakis, P., White, W., and Tombach, I. 2000. Guidance for the performance evaluation of three-dimensional air quality modeling systems for particulate matter and visibility. *J. Air Waste Manage. Assoc*. 50:588–599.

Seigneur, C., and Moran, M. D. 2004. Using models to estimate particle concentration. In *Particulate matter science for policy makers: A NARSTO assessment*, eds. P. McMurry, M. Shepherd, and J. Vickery, chapter 8. Cambridge: Cambridge University Press.

Seilkop, S. K. 1995a. Representativeness of surface site air concentrations relative to an 80 km grid. In *Regional photochemical measurement and modeling studies. Vol. 1. Results and interpretation of field measurements*, eds. A. J. Ranzieri and P. A. Solomon, pp. 197–210. Pittsburgh, PA: Air & Waste Management Association.

Seilkop, S. K. 1995b. *Characterization and usage of subgrid scale variability in model evaluation*. Report prepared by Analytical Services, Inc., Durham, NC, for U.S. Environmental Protection Agency, Research Triangle Park, NC.

Seinfeld, J. H., and Pandis, S. N. 1998. *Atmospheric chemistry and physics—From air pollution to climate change*. New York: John Wiley & Sons.

Setton, E., Hystad, P., Keller, C. P., Cloutier-Fisher, D., Foster, L., Copes, R., and Brauer, M. Personal communication. Simulating risk of exposure to traffic-related air pollution in urban and suburban working and non-working populations. Presented at Air Quality 2006: Piecing the Puzzle. The Fourth Canadian Workshop on Air Quality. Feb. 15, 2006. Banff, AB.

Sillman, S., He, D., Cardelino, C., and Imhoff, R. E. 1997. The use of photochemical indicators to evaluate ozone–NOx–hydrocarbon sensitivity: Case studies from Atlanta, New York, and Los Angeles. *J. Air Waste Manage. Assoc.* 47:1030–1040.

Sillman, S., He, D., Pippin, M. R., Daum, P.H., Imre, D. G., Kleinman, L. I., Lee, J. H., and Weinstein-Lloyd, J. 1998. Model correlations for ozone, reactive nitrogen and peroxides for Nashville in comparison with measurements: Implications for $O_3$–$NO_x$ hydrocarbon chemistry. *J. Geophys. Res. Atmos*. 103:22629–22644.

Silva, C., Gould, T., Larson, T., Henderson, S., Aleen R., Cohen, M., and Buzzelli, M. Personal communication. A tale of two cities: Comparing land use and traffic related pollution exposure models in Seattle and Vancouver. Presented at Air Quality 2006: Piecing the Puzzle. The Fourth Canadian Workshop on Air Quality. Feb. 15, 2006. Banff, AB.

Smith, K. R. 1988. Air pollution: Assessing total exposure in the United States. *Environment* 30:10.

Smyth, S. C., Jiang, W., Yin, D., Roth, H., and Giroux, E. 2006a. Evaluation of CMAQ $O_3$ and $PM_{2.5}$ performance using Pacific 2001 measurement data. *Atmos. Environ*. 40:2735–2749.

Smyth, S., Yin, D., Roth, H., Jiang, W., Moran, M. D., and Crevier, L.-P. 2006b. The impact of GEM and MM5 meteorology on CMAQ air quality modeling results in eastern Canada and the northeastern United States. *J. Appl. Meteorol*. 45:1525–1541.

Spengler, J. D., Koutrakis, P., Dockery, D. W., Raizenne, M., and Speizer, F. E. 1996. Health effects of acid aerosols on North American children: Air pollution exposures. *Environ. Health Perspect.* 104:492–499.

Stedman, J. 2004. The predicted number of air pollution related deaths in the UK during the August 2003 heatwave. *Atmos. Environ.* 38:1083–1085.

Stein, A. F., and Lamb, D. 2002. Chemical indicators of sulfate sensitivity to nitrogen oxides and volatile organic compounds. *J. Geophys. Res. Atmos.* 107:13-1–13–11.

Stern, N. 2007. *The economics of climate change*. London: HM Treasury. Available at http://www.hm-treasury.gov.uk/independent_reviews/stern_review_economics_climate_change/stern_review_report.cfm.

Stieb, D. M., Paola, J., and Neuman, K. 1996. Do smog advisories work? Results of an evaluation of the Canadian smog advisory program. *Can. J. Public Health* 87:166–169.

Stieb, D. M., Judek, S., and Burnett, R. T. 2002. Meta-analysis of time-series studies of air pollution and mortality: Effects of gases and particles and the influence of cause of death, age, and season. *J. Air Waste Manage. Assoc.* 52:470–484.

Stieb, D. M., Judek S., and Burnett R. T. 2003. Meta-analysis of time-series studies of air pollution and mortality: Update in relation to the use of generalized additive models. *J. Air Waste Manage. Assoc*. 53:258–261.

Streets, D. G., Yu, C., Bergin, M. H., Wang, X., and Carmichael, G. R. 2006. Modeling study of air pollution due to the manufacture of export goods in China's Pearl River Delta. *Environ. Sci. Technol.* 40:2099–2107.

Sustainable Development Technology Canada. 2007. *SDTC funding breakdown by sector*. Available at http://www.sdtc.ca/en/results/portfolio/economic_sectors.htm.

Thurston, G. D., Ito, K., Hayes, C. G., Bates, D., and Lippmann, M. 1994. Respiratory hospital admissions and summertime haze air pollution in

Toronto, Ontario: Consideration of the role of acid aerosols. *Environ. Res.* 65:271–290.

Toronto Public Health. 2000. *Air pollution burden of illness in Toronto*. Toronto, Ontario. Available at www.toronto.ca/health/burdenof.pdf.

Toronto Public Health. 2001. Condition critical: Fixing our smog warning system. Toronto, Ontario. Available at www.toronto.ca/health/hphe/pdf/report_condition_critical_technical.pdf.

Toronto Public Health. 2007. *Air Pollution Burden of Illness from Traffic in Toronto – Problems and Solutions*. Toronto, Canada. Available at http://www.toronto.ca/health/hphe.

Uliasz, M. 1993. The atmospheric mesoscale dispersion modeling system. *J. Appl. Meteor*. 32:139–149.

U.S. Environmental Protection Agency. 2001. *Guidance for demonstrating attainment of air quality goals for $PM_{2.5}$ and regional haze*. U.S. Environmental Protection Agency Report Draft 2.1. Available at www.epa.gov/scram001/guidance/guide/draft_pm.pdf.

U.S. Environmental Protection Agency. 2003. *National air quality and emissions trends report, 2003 special studies edition*. U.S. Environmental Protection Agency. Available at http://nsdi.epa.gov/air/airtrends/aqtrnd03/pdfs/toc_figs_tables.pdf.

U.S. Environmental Protection Agency. 2004. *Air Quality Criteria Document for Particulate Matter. Volume II*. EPA/600/P-99/002bF. Research Triangle Park, NC: Office of Research and Development, U.S. EPA. Available at http://cfpub2.epa.gov/ncea/cfm/recordisplay.cfm?deid=87903.

U.S. Environmental Protection Agency. 2005a. *Review of the national ambient air quality standards for particulate matter: Policy assessment of scientific and technical information*. OAQPS staff paper. Available at www.epa.gov/ttn/naaqs/standards/pm/data/pmstaffpaper_20051221.pdf.

U.S. Environmental Protection Agency. 2005b. *Technical support document for the final clean air interstate rule air quality modeling*. U.S. Environmental Protection Agency. Available at www.epa.gov/cair/pdfs/finaltech02.pdf.

U.S. Environmental Protection Agency. 2005c. *Guidance on the use of models and other analyses in attainment demonstrations for the 8-hour ozone NAAQS*. EPA-454/R-05-002. U.S. Environmental Protection Agency. Available at www.epa.gov/scram001/guidance/guide/8-hour-o3-guidance-final-version.pdf.

U.S. Environmental Protection Agency. 2006a. *Provisional assessment of recent studies on health effects of particulate matter exposure.* EPA/600/R-06/063. Research Triangle Park, NC: National Center for Environmental Assessment. Available at http://epa.gov/pm/pdfs/ord_report_20060720.pdf.

U.S. Environmental Protection Agency. 2006b. *Environmental justice*. U.S. Environmental Protection Agency. Available at http://www.epa.gov/compliance/environmentaljustice/index.html.

Venkatram, A. 1988. Inherent uncertainty in air quality modeling. *Atmos. Environ*. 22:1221–1227.

Watson, J. G. 2002. Visibility: Science and regulation. *J. Air Waste Manage. Assoc.* 52:628–713.

Watson, J. G., and Chow, J. C. 2005. Chapter 16B Receptor Models. In *Air quality modeling: Theories, methodologies, computational techniques, and available databases and software, Vol. II*, ed. P. Zannetti, pp. 455–501. EnviroComp Institute and Air and Waste Management Association.

Watson, J. G., Chow, J. C., and Fujita, E. M. 2001. Review of volatile organic compound source apportionment by chemical mass balance. *Atmos. Environ.* 35:1567–1584.

West, J. J., Ansari, A. S., and Pandis, S. N. 1999. Marginal $PM_{2.5}$: Nonlinear aerosol mass response to sulfate reductions in the eastern United States. *J. Air Waste Manage. Assoc*. 49:1415–1424.

Westerdahl, D., Fruin, S., Sax, T., Fine, P. M., and Sioutas, C. 2005. Mobile platform measurements of ultrafine particles and associated pollutant concentrations on freeways and residential streets in Los Angeles. *Atmos. Environ*. 39:3597–3610.

Wexler A. S., and Johnson M. V. 2006. What have we learned from highly time resolved measurements during the EPA supersite program and related studies? *J. Air Waste Manage. Assoc*. Accepted.

Wheeler, A. J., Smith-Doiron, M., Xu, X., Gilbert, M. L., and Brook, J. R. 2008. Intra-Urban variability of air pollution in Windsor, Ontario–Measurement and modeling for human exposure assessment. *Environmental Research.*106:7–16.

Wheeler, A. J., Xu, X., Smith-Doiron, M., Gilbert, N. L., and Brook, J. R. Personal communication. Intra-urban variability of air pollution – monitoring and modeling of PM2.5, PM2.5-10, and NO2. Poster presentation at ISEE ISEA 2006, Paris.

Williams, M. L. 2008. Air quality risk management. *J Toxicol Environ Health* 71:9–12.

Williams, M. L. 2007. UK Air quality in 2050—Synergies with climate change policies, *Environ. Sci. Policy* 10:169–175.

Wilson, W. E., and Brauer, M. 2006. Estimation of ambient and non-ambient components of particulate matter exposure from a personal monitoring panel study. *J. Expo. Sci. Environ. Epidemiol.* 16:264–274.

Winer, A. M., Lurmann, F. W., Coyner, L. A., Colome, and Poe, M. P. 1989. *Characterization of air pollutant in the California south coast air basin: Application new regional human exposure (REHEX) model*. Report to the South Coast air quality management district and the California state university foundation.

Wong, C. M., Hu, Z. G., Lam, T. H., Hedley, A. J., and Peters, J. 1999. Effects of ambient air pollution and environmental tobacco smoke on respiratory health of non-smoking women in Hong Kong. *Int J Epidemiol* 28:859–864.

Wong, C. M., Lam., T. H., Peters, J., Hedley, A. J., Ong, S. G., Tam, A. Y. C, Liu, J., and Spiegelhalter D. J. 1998. Comparison between two districts of the effects of an air pollution intervention on brochial responsiveness in primary school children in Hong Kong. *J. Epidemiol. Commun. Health* 52:571–578.

World Health Organization. 1999. *Monitoring ambient air quality for health impact assessment*. WHO Regional Publications, European Series, No. 85. Copenhagen, Denmark.

World Health Organization. 2000. *Air quality guidelines for Europe*. WHO Regional Publications, European Series, No. 91. Copenhagen, Denmark.

World Health Organization. 2002. *The health effects of indoor air pollution exposure in developing countries*. WHO/SDE/OEH/02.05. Available at http://whqlibdoc.who.int/hq/2002/WHO_SDE_OEH_02.05.pdf.

World Health Organization. 2004. *Meta-analysis of time-series studies and panel studies of particulate matter (PM) and ozone (O3)*. Copenhagen: WHO Regional Office for Europe. Available at www.euro.who.int/document/E82792.pdf.

World Health Organization. 2005a. *WHO air quality guidelines global update 2005*. Report on a working group meeting, Bonn, Germany. Available at http://www.euro.who.int/Document/E87950.pdf.

World Health Organization. 2005b. *Effects of air pollution on children's health and development. A review of the evidence*. Available at www.euro.who.int/document/E86575.pdf.

World Health Organization. 2005c. *Health effects of transport-related air pollution,* eds. M. Krzyzanowski, B. Kuna-Dibbert, and J. Schneider, pp. 1–206. Copenhagen, Denmark: WHO Regional Office for Europe. Available at www.euro.who.int/document/e86650.pdf.

World Health Organization. 2006a. *Health risks of particulate matter from long-range transboundary air pollution.* Available at www.euro.who.int/document/E88189.pdf.

World Health Organization. 2006b. *Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide. Global update 2005. Summary of risk assessment*. Geneva: World Health Organization. Available at www.euro.who.int/air/activities/20050222_2.

Wu, J., Lurmann, F., Winer, A., Lu, R., Turco, R., and Funk, T. 2005. Development of an individual exposure model for application to the Southern California children's health study. *Atmos. Environ*. 39: 259–273.

Xu, X., Brook, J. R., and Guo, P. 2007. A statistical assessment of saturation and mobile sampling strategies to estimate long term average concentrations across urban areas. *J. Air Waste Man. Assoc*. 57:1396–1406.

Yap, D., Reid, N., De Brou, G., Bloxam, R., Piché, E., Chan, W., Cheng, C., Bitzos, M., Wong, S., D. S. Harper Consulting, DSS Management Consultants Inc., and Potvin Air Management Consulting. 2005. *Transboundary air pollution in Ontario*. Ontario Ministry of the Environment, Toronto, Ontario. Available at www.ene.gov.on.ca/envision/techdocs/5158e.pdf.

Yli-Tuomi, T., Aarnio, P., Pirjola, L., Mäkelä, T., Hillamo, R., and Jantunen, M. 2005. Emissions of fine particles, $NO_x$, and CO from on-road vehicles in Finland. *Atmos. Environ*. 39:6696–6667.

Yu, S., Dennis, R., Roselle, S., Nenes, A., Walker, J., Eder, B., Schere, K., Swall, J., and Robarge, W. 2005. An assessment of the ability of three-dimensional air quality models with current thermodynamic equilibrium models to predict aerosol $NO_3^-$. *J. Geophys. Res. Atmos.* 110:D07S13–1–D07S13-22.

Yu, S., Dennis, R., Bhave, P. V., and Eder, B. K. 2004. Primary and secondary organic aerosols over the United States: estimates on the basis of observed organic carbon (OC) and elemental carbon (EC), and air quality modeled primary OC/EC ratios. *Atmos. Environ*. 38:5257–5268.

Zanobetti, A., Schwartz, J., and Dockery, D. W. 2000a. Airborne particles are a risk factor for hospital admissions for heart and lung disease. *Environ. Health Perspect.* 108:1071–1077.

Zanobetti, A., Wand, M. P., Schwartz, J., and Ryan, L. M. 2000b Generalized additive distributed lag models: Quantifying mortality displacement. *Biostatistics* 1:279–292.

Zanobetti, A., and Schwartz, J. 2001. Are diabetics more susceptible to the health effects of airborne particles? *Am. J. Respir. Crit. Care Med*. 164: 831–833.

Zanobetti, A., and Schwartz, J. 2003. Multicity assessment of mortality displacement within the APHEA2 project. In: *Revised analyses of time-series studies of air pollution and health*, pp. 249–254. Boston: Health Effects Institute. Available at http://pubs.healtheffects.org/getfile.php?u=21.

Zeger, S. L., Dominici, F., and Samet, J. 1999. Harvesting-resistant estimates of air pollution effects on mortality. *Epidemiology* 10:171–175.

Zhang, Y., Seigneur, C., Seinfeld, J. H., Jacobson, M. Z., Clegg, S. L., and Binkowski, F. S. 2000. A comparative review of inorganic aerosol thermodynamic equilibrium modules: Similarities, differences, and their likely causes. *Atmos. Environ*. 34:117–137.

Zhang, L., Moran, M. D., and Brook, J. R. 2001. A comparison of models to estimate in-canopy photosynthetically active radiation and their influence on canopy stomatal resistance. *Atmos. Environ*. 35:4463–4470.

Zhang, L., Moran, M. D., Makar, P. A., Brook, J. R., and Gong, S. 2002. Modelling gaseous dry deposition in AURAMS—A unified regional air-quality modelling system. *Atmos.. Environ*. 36:537–560.

Zhang, Y., Vijayaraghavan, K., and Seigneur, C. 2005. Evaluation of three probing techniques in a three-dimensional air quality model. *J. Geophys. Res. Atmos*. 110:1–21.

Zuber, A., and Keating, T. J. 2005. *Convention on long-range transboundary air pollution: Task force on hemispheric transport of air pollution status and outlook—December 2005*. Available at www.unece.org/env/documents/2005/eb/EB/Inf.Doc.05.Hemispheric%20Transport.pdf.

Zunckel, M., Koosailee, A., Yarwood, G., Maure, G., Venjonoka, K., van Tienhoven, A.M., and Otter, L. 2006. Modelled surface ozone over southern Africa during the Cross Border Air Pollution Impact Assessment Project. *Environ. Model. Software* 21:911–924.